AN ECOLOGICAL-INTEGRITY ASSESSMENT OF THE NEW JERSEY PINELANDS

A COMPREHENSIVE ASSESSMENT OF THE LANDSCAPE AND AQUATIC AND WETLAND SYSTEMS OF THE REGION

PINELANDS COMMISSION
APRIL 2008

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A COMPREHENSIVE ASSESSMENT OF THE LANDSCAPE AND AQUATIC AND WETLAND SYSTEMS OF THE REGION

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April 2008

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PREFACE

The Pinelands Comprehensive Management Plan (CMP, Pinelands Commission 1980) was adopted in 1980 following a planning process that began in 1979. What sets the CMP apart from most other land-use regulation and planning programs is that it was based on an ecosystem approach to protecting the natural resources of the New Jersey Pinelands (Figures 1 and 2, Good and Good 1984, Robichaud-Collins and Russell 1988, Russell 1994, Walker and Solecki 1999). A major element of the CMP is a land-use-management program that directs development away from areas considered ecologically critical to areas deemed less critical.

Implementation of the CMP over the past three decades has involved more local and site-specific approaches. At times, single issues have overshadowed the broader, multifaceted goal of ensuring the long-term preservation of a unique ecosystem. The purpose of this ecological-integrity assessment is to focus on the big picture again by evaluating the current ecological status of the entire 938,173-acre (379,827-ha) Pinelands Area and the ecosystem that it represents.

The concept of ecological integrity as it was applied to the Pinelands Area, a description of the approach used to characterize ecological integrity, and the results of the assessment are presented in the main body of this report. The basis and background for the ecological principles and methods used in the assessment are reviewed in Appendix 1. Additional supporting documentation is provided in Appendices 2 through 4.

Results of the assessment can be used to evaluate current Pinelands management-area and zoning designations, identify areas best suited for clustering development, and provide a regional basis for the review of individual projects. Other possible applications include development of habitat-conservation plans, the identification of important areas for acquisition, preparation of cumulative-watershed-impact assessments, and, as presented in Appendix 5, a comprehensive evaluation of wetland integrity throughout the Pinelands. The methodology used in the assessment also provides the means to reevaluate the ecological integrity of the Pinelands as new data become available.

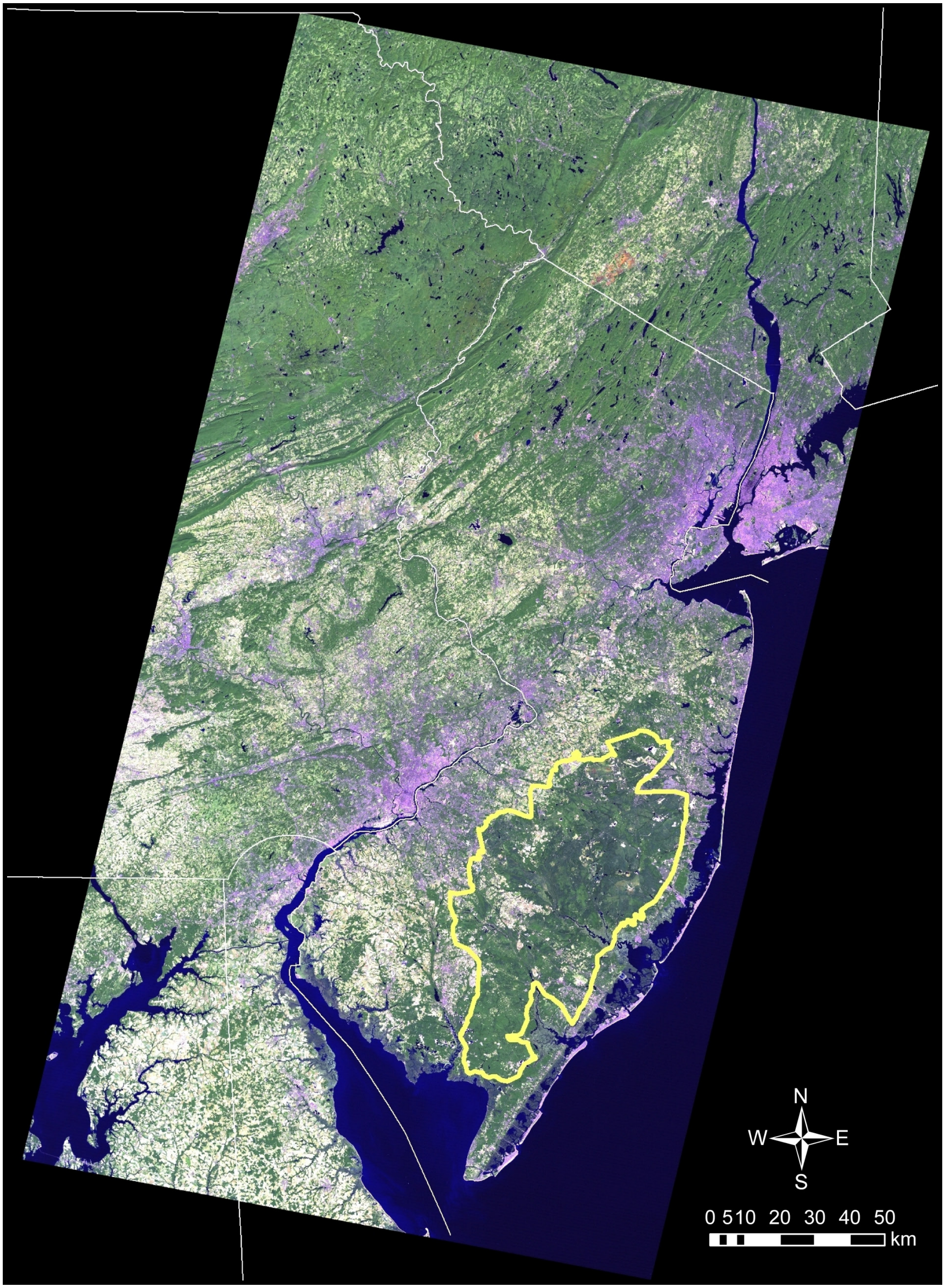

Figure 1. Landsat image of the Pinelands. The boundary of the Pinelands Area is shown in yellow. Landsat image source: U. S. Geological Survey.

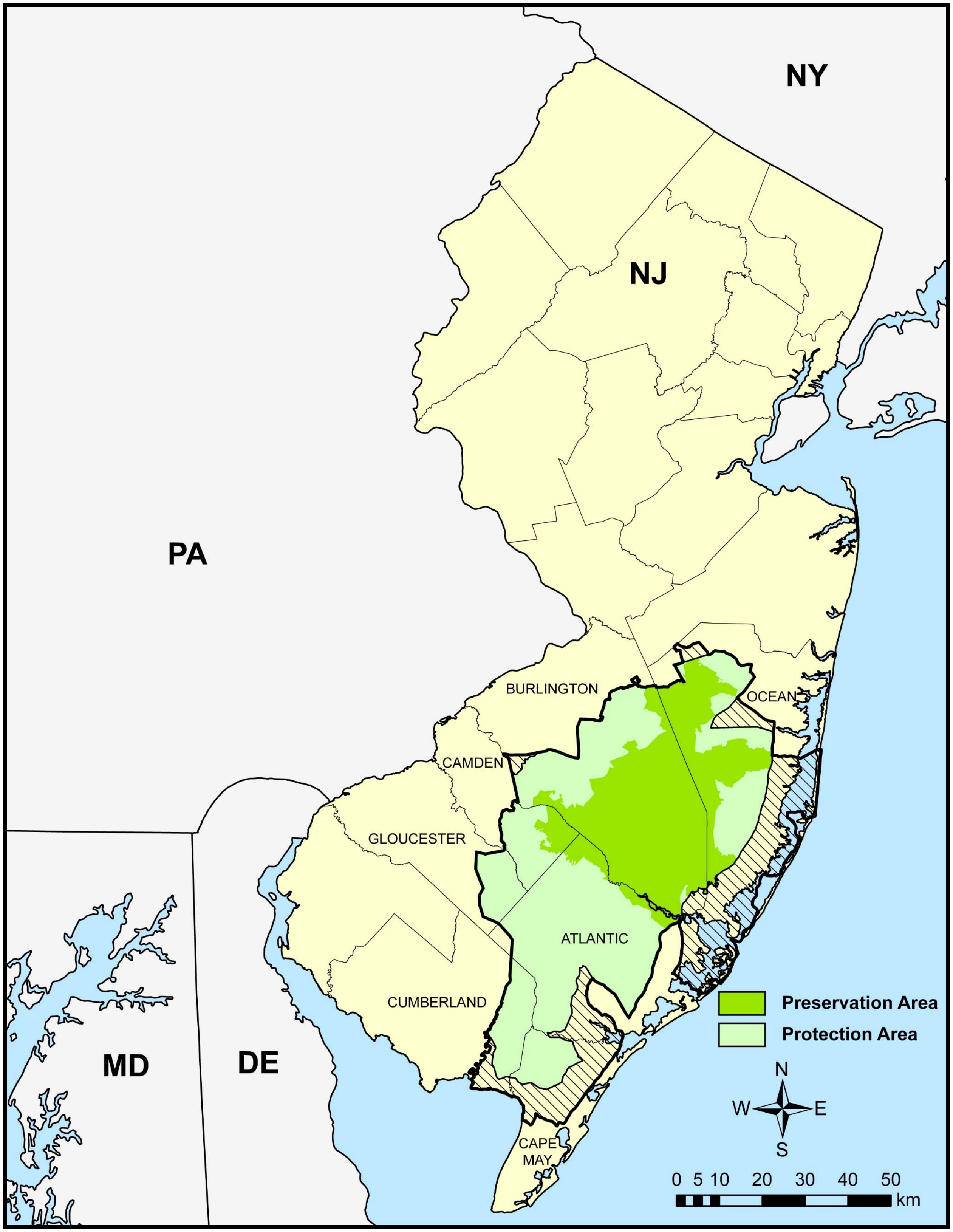

Figure 2. The New Jersey Pinelands National Reserve and Pinelands Area. The Pinelands Area includes the Preservation Area and the Protection Area. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

ACKNOWLEDGMENTS

We thank members of the Pinelands Commission's Science Advisory Committee, including Richard Lathrop, Joan Ehrenfeld, John Dighton, Anthony Navoy, and Peter Oudemans, whose comments and suggestions helped to improve the final assessment. We also appreciate the comments provided by Jonathan Kennen and Robert Nicholson. David Golden and David Jenkins provided data on the distribution of threatened and endangered animal species. Funding for the project was provided through the Pinelands Conservation Fund.

ECOLOGICAL INTEGRITY CONCEPTS AND PRINCIPLES

What is Ecological Integrity?

Leopold (1949), who stated “A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends to do otherwise,” is credited as the first person to apply the term integrity to the natural world (Noss 1990, Haynes et al. 1996, Andreasen et al. 2001, Quigley et al. 2001). This application is reflected in the Clean Water Act (Water Pollution Control Act Amendments of 1972), which addresses the need to "restore and maintain the chemical, physical and biological integrity of the Nation's waters" (Ballentine and Guarraia 1977). Building on definitions presented in Ballantine and Guarraia (1977), Karr and Dudley (1981) defined ecological integrity as “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a specific composition, diversity, and functional organization comparable to that of natural habitat of a region,” noting that this definition does not imply beneficial use of a natural resource.

Karr and Dudley (1981) equated ecological integrity with the summation of chemical, physical, and biological integrity and associated it with ecosystems that can withstand and recover from most natural and many human disturbances. The concept of integrity has been most frequently applied to aquatic systems using fish, macroinvertebrates, periphyton, and vegetation as biological indices (Angermeier and Karr 1986, Blocksom et al. 2002, Drake and Valley 2005, Fausch et al. 1990, Griffith et al. 2005, Hill et al. 2003, Karr 1981, 1991, Klemm et al. 2003, Miller et al. 1988, Simon 1999, 2003, Stewart et al. 2003, among many others). In this regard, Allan (2004) defined ecological integrity as a term that describes the status of stream ecosystems and their responses to human influences in relation to least-impaired reference sites.

Carignan and Villard (2002) and Parrish et al. (2003) adapted Karr and Dudley’s (1981) definition of ecological integrity. Like Karr and Dudley (1981), Parrish et al. (2003) referred to the natural habitats in a region when defining ecological integrity, whereas Carignan and Villard (2002) used the term undisturbed rather than natural in their definition. The term natural is difficult to define (Andreasen et al. 2001). This is especially true in the Pinelands, where few areas have been unaffected by human interference in natural processes (Wacker 1979). Eighteenth, nineteenth, and early twentieth century resource exploitation, including timber harvesting, charcoal making, mining, damming of streams, iron smelting, iron and glass manufacturing, and agriculture have left a visible mark on the landscape. The devastation wrought by this early exploitation has contributed substantially to the creation of many present-day habitats and landscape patterns considered characteristic of the region.

Regier (1993) associated ecological integrity with the ability of a system to recover toward an end state that is normal or “good” for that system, but not necessarily pristine. Haynes et al. (1996) suggested that because measures of integrity or resiliency require judgments based on comparisons of subjectively chosen indicators, ecological integrity is more an expression of policy than science. Expressions of policy may, in effect, imply beneficial use of a natural resource, which appears contrary to the opinion of Karr and Dudley (1981).

In the New Jersey Pinelands, ecological integrity can be defined within the context of a long history of scientific research that has documented the “natural” character of the ecosystem over the past century, the Pinelands Comprehensive Management Plan (Pinelands Commission 1980), and the Plan’s enabling legislation, thus reflecting both science and policy. The extensive scientific literature on Pinelands ecology provides a sound basis for characterizing water-quality conditions, determining with a fairly high level of confidence which plant and animal species are native to the region, and describing the present-day characteristics of Pinelands habitats. The scope of the research conducted in the Pinelands through the latter part of the 20th century is well documented in Forman (1979a), Buchholz and Good (1982), and Gemmell et al. (1989).

Because present-day species composition and habitat structure reflect an ecosystem recovering from the intense resource exploitation of past centuries, one can only speculate about natural, pre-European conditions. However, the natural-resource-policy statements included in the Pinelands Protection Act and the Pinelands Comprehensive Management Plan provide a reference for determining what is "natural."

The natural-resources goal of the Pinelands Comprehensive Management Plan is to "preserve, protect, and enhance the overall ecological values of the Pinelands, including its large forested areas, its essential character, and its potential to recover from disturbance." This goal was developed in response to the National Parks and Recreation Act of 1978 and the Pinelands Protection Act, which included goals for the Comprehensive Management Plan with respect to the outer Protection Area and the inner Preservation Area (Figure 2).

Among the goals set for the Protection Area by the legislation are the preservation and maintenance of the essential character of the existing Pinelands environment, which includes the indigenous plant and animal species and their habitats, and the protection and maintenance of surface water and groundwater quality. The goals for the Preservation Area include the preservation of an extensive and continuous area of land in its natural state, insuring the continuation of a Pinelands environment and the region's unique and significant ecological and other resources, protecting and preserving the quantity and quality of existing surface water and groundwaters, and providing a sufficient amount of undeveloped land to accommodate management practices, such as controlled fire, necessary to maintain the special ecology of the core area. Both the state and federal Pinelands legislation also directed the Pinelands Commission to conduct a resource assessment that addressed the overall ecological values of the Pinelands and the factors affecting the "ecological integrity" of the region.

The references to indigenous plant and animal species and their habitats, the protection and maintenance of water quality, the potential to recover from disturbance, and ecosystem processes such as fire suggest that Karr and Dudley's (1981) definition of ecological integrity can be applied in an ecological-integrity assessment of the Pinelands. Natural Pinelands habitats may be equated with existing upland forests, wetland landscapes, and aquatic systems that provide habitat for indigenous plants and animals. This definition of natural is consistent with that given by Forman (1995), who described natural vegetation as the plant-species composition and cover of an area that has not been planted by humans, noting that although human impacts and exotic species are often present, native species usually dominate the area of natural vegetation.

The goals of the Pinelands legislation are similar to the ecosystem-management goals described in a widely cited paper by Grumbine (1994). Five specific goals presented by Grumbine (1994) include: 1) maintaining viable populations of all native species in place; 2) protecting native ecosystem types; 3) maintaining processes such as disturbance regimes, hydrologic processes, and nutrient cycles; 4) long-term management; and 5) the accommodation of human use within the constraints established by the other four goals.

Spatial and Temporal Scales and Ecological Hierarchies

Different aspects of ecological integrity may become apparent at different spatial and temporal scales and levels in an ecological hierarchy (Noon and Dale 2002, De Leo and Levin 1997). Turner et al. (2001) defined scale as the "spatial or temporal dimensions of an object or process, characterized by both grain and extent," where grain is the "finest level of spatial resolution possible within a given data set." Changing grain size and extent can affect landscape metrics such as the number of patches, patch size, and edge (Wu et al. 2002). Ecological hierarchy refers to the levels of organization that include genes, individuals, populations, species, communities, ecosystems, and landscapes, where the higher levels in the hierarchy constrain and control the lower levels to various degrees (Noon and Dale 2002, Noss 1990, Turner et al. 2001). The levels of organization in a stream hierarchy range from watersheds to successively smaller scales that include channel reaches, riffles, pools, and microhabitats (Allan 2004). Spatial scales used to relate stream conditions to land use include a local stream reach, an entire stream with an adjacent terrestrial buffer, or an entire watershed (Morley and Karr 2002, Allan 2004). Different processes characterize each level of organization in the ecological hierarchy, each with their own spatial and temporal scales.

Interacting systems of wetlands at the watershed, landscape, and regional level represent the appropriate scale for wetland assessments (Bedford and Preston 1988). Likewise, Naiman et al. (1993) suggested that a landscape perspective is needed to maintain species and ecological processes in riparian systems. This approach is especially important when regulating the cumulative impact of incremental wetland losses (Lee and Gosselink 1988, Gosselink et al. 1990).

Noss (1990) described two opposing trends in conservation ecology, with one emphasizing individual species and another emphasizing whole communities, ecosystems, and landscapes. Grand et al. (2004) used landscape variables to predict rare and declining bird- and moth-species hotspots in a southeastern Massachusetts pitch pine–scrub oak (*Pinus rigida-Quercus ilicifolia*) community. Based on the results, they suggested that the surrounding landscape influenced habitat suitability for both moths and birds. Because few bird and moth hotspots overlapped, they concluded that protecting habitat for one taxon would not protect the other and suggested that multi-taxa, multi-scale approaches should be used in pitch pine–scrub oak communities. Freemark et al. (2002a) also indicated that conservation at multiple spatial and temporal scales might be more efficient than focusing on single species. Maddock and du Plessis (1999) supported this position and listed several potential problems with relying on species data only, such as the age of data points in a rapidly changing landscape, uneven sampling distribution across the landscape, and coverage that is representative of only a small portion of species. Emphasis on species-based approaches rather than ecosystem- and landscape-level approaches may not conserve the majority of existing biological diversity in a region (Franklin 1993, Maddock and du Plessis 1999).

Protecting the majority of species in an area depends on the scale at which ecological integrity is assessed (Noon and Dale 2002). For example, Steffan-Dewenter et al. (2002) concluded that the importance of the landscape context for local-pollinator communities could only be detected by analysis of multiple scales. Species richness and abundance of solitary wild bees in a grassy-field margin adjacent to cereal fields was related to the percentage of semi-natural habitats within 750 m (2,461 ft), whereas honey bees were correlated with landscape context at larger scales up to 3,000 m (9,842 ft). They suggested that because most ecological processes depend on scales larger than a single habitat, spatial patterns and ecological processes should be linked at a landscape scale.

Conservation efforts frequently focus on the protection of rare species. The role of rare species in ecosystem management was discussed at a 1982 conference concerning ecological solutions to environmental concerns in the Pinelands (Good 1982). Scientists participating in a session on ecosystem fragmentation[1] concluded that rare species are not likely to be good ecosystem indicators and that managing for rare species is not the same as managing for ecosystem persistence. King (1993) expressed a similar opinion, noting that because they are rare, rare species are unlikely to affect ecosystem function to a great extent. A practical reason for not using rare species as the

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[1] Fragmentation session participants: R. G. Risser, R. L. Burgess, R. T. T. Forman, J. Terborgh, J. A. Wiens, and R. A. Zampella

basis for ecosystem management is that rare-species data are frequently treated as confidential and are not available for public or scientific scrutiny. Also, the ability to predict the occurrence of individual rare species from habitat data is limited by a lack of data needed to adequately characterize habitat and, because the species are rare, the reduced statistical power of small-sample sizes used to develop habitat-based models (Wiser et al. 1998).

Good and Good (1984) emphasized the importance of incorporating an ecosystem approach for the preservation of a significant portion of the Pinelands. Referring to the Pinelands, Whittaker (1979) noted that “Conservation should seek the preservation of landscapes of communities interrelated by topography and soil gradients, movement of water and soil nutrients, and dispersal of plant and animal populations.” He indicated that the Pinelands should be preserved as a landscape pattern and commented that conservation efforts are not adequately served by preserving individual pieces of the landscape representing community types.

Ecological-integrity Principles

The Pinelands ecological-integrity assessment was guided by three basic principles concerning landscape, aquatic, and wetland-drainage integrity. The principles were based on the results of ecological studies conducted in the Pinelands and elsewhere (reviewed in Appendix 1). Landscape integrity focuses on species that move across wetlands and uplands and processes that operate at a regional-landscape level. Aquatic integrity deals primarily with processes that operate at the watershed level and the species and communities that are influenced by the quantity and quality of surface waters. Wetland-drainage-integrity focuses on upland land uses that affect the quantity and quality of groundwater flowing to palustrine wetlands. These three integrity measures were used to determine the overall ecological integrity of the 938,173-acre (379,827-ha) Pinelands Area.

Landscape Integrity. *Conservation of characteristic Pinelands plant and animal species and communities, including wide-ranging species, requires the protection of relatively large tracts of Pinelands habitat, including upland forests, wetlands, and water bodies.*

Aquatic Integrity. *Conservation of characteristic Pinelands water quality and lake, pond, and stream communities and the indigenous plant and animal species that make up these communities requires the protection of associated watersheds.*

Wetland-drainage Integrity. *Conservation of characteristic Pinelands palustrine wetlands and the indigenous plant and animal species that inhabit these wetlands requires the protection of adjacent uplands that influence the hydrologic integrity of the wetlands.*

Ecological Integrity. *The overall ecological integrity of the Pinelands is a composite of landscape-, watershed-, and wetland-drainage-integrity measures.*

EVALUATING PINELANDS ECOLOGICAL INTEGRITY

The Pinelands ecological-integrity assessment was conducted at three levels of an ecological hierarchy, including the entire regional upland-forest and wetland landscape, aquatic systems and associated watersheds, and freshwater wetlands and adjacent upland areas. The scale at which the assessment of each level was conducted varied in both extent and grain. As suggested by Gutzwiller (2002), each assessment treated upland and wetland systems as inseparable.

Landscape integrity, aquatic integrity, wetland-drainage integrity, and overall Pinelands ecological integrity were determined for the entire Pinelands Area and for each Pinelands management area. The Pinelands Comprehensive Management Plan established nine management areas with varying permitted land-use and development intensities (Table 1, Figure 3). The Pinelands Commission deemed the Preservation Area District the most critical ecological region in the Pinelands. Development is severely restricted in this management area. The overall ecological value of Forest Areas, where the type and intensity of permitted land uses are also restricted, is considered similar to the Preservation Area District. Special Agricultural Production Areas are used primarily for berry agriculture and horticulture of native Pinelands plants. Development in Special Agricultural Production Areas is generally limited to farm-related uses. Military and Federal Installation Areas include federal facilities such as Fort Dix and McGuire Air Force Base. Agricultural Production Areas are active upland-agricultural lands and adjacent areas with suitable agricultural soils. Permitted uses in the Agricultural Production Areas include both farm-related and non-farm housing, with the latter requiring larger lots sizes. Limited, low-density residential development and roadside retail are permitted in Rural Development Areas. The most intensive type and level of residential and commercial uses are permitted in Pinelands Villages, Pinelands Towns, and Regional Growth Areas.

The focus of the Pinelands ecological-integrity assessment was the status of the Pinelands Area in 2002, the period for which the most current and detailed land-use data were available. To identify areas of potentially high ecological integrity that were affected by land-use changes and to provide an indication of the cumulative effect of land-use changes in one area on adjacent areas, assessments were also completed using 1986 and 1995 land-use data. Although the 2002 land-use/land-cover data represent the most current and detailed information on the status of the Pinelands landscape, substantial changes have occurred in some rapidly developing Regional Growth Areas in municipalities such as Stafford Township, Hamilton Township, and Egg Harbor Township, and extensive areas of cranberry bogs are no longer actively farmed in the vicinity of Chatsworth, Woodland Township. Real-estate parcel data, which show land-use commitments at the individual-lot level, can prove useful when planning for areas where major changes are known to have occurred. Assessing the detailed effect of these changes using the methodology applied to the 1986, 1995, and 2002 periods cannot be done until land-use/land-cover data for the recent 2007 aerial-photograph coverage are prepared.

Table 1. Pinelands management areas. The area and the percentage of the Pinelands Area (%) that it represents are given for each management area.

Management area	Code	Acres	Hectares	%
Regional Growth Area	RGA	76,472	30,960	8.2
Pinelands Town	PT	21,758	8,809	2.3
Pinelands Village	PV	25,907	10,489	2.8
Rural Development Area	RDA	113,181	45,822	12.1
Federal and Military Installation Area	FMI	47,550	19,251	5.1
Agricultural Production Area	APA	68,160	27,595	7.3
Special Agricultural Production Area	SAP	37,582	15,215	4.0
Forest Area	FA	252,950	102,409	27.0
Preservation Area District	PrA	294,612	119,276	31.4
Pinelands Area	PA	938,173	379,827	100.0

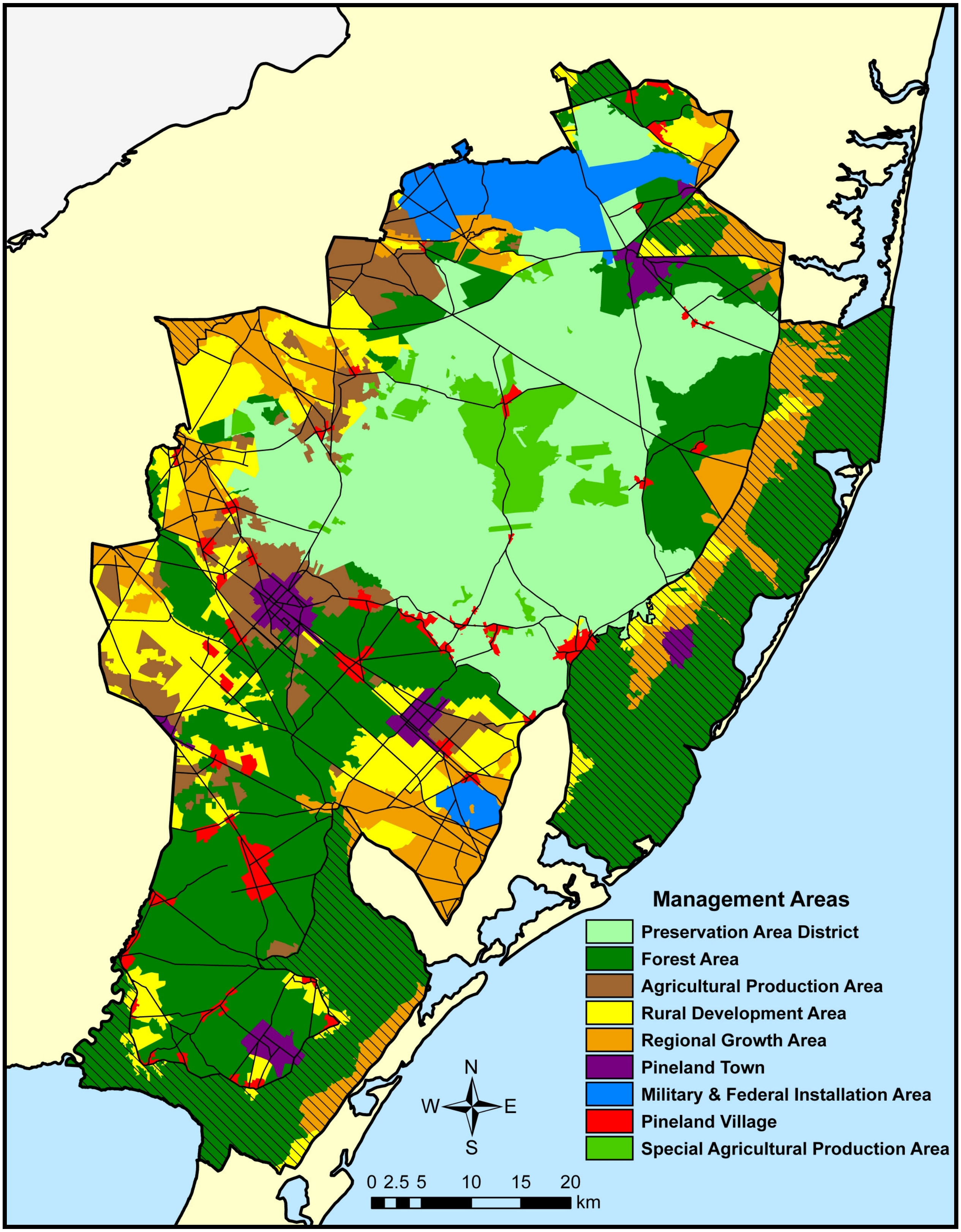

Figure 3. Pinelands management areas. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

Landscape Integrity

Definition: Landscape integrity is a measure of the extent of Pinelands habitat in an area. Pinelands habitat includes upland forests, water, and wetlands, with the exception of managed wetlands and wetland agriculture.

Background. The review of the landscape-ecology literature presented in Appendix 1 indicated that habitat-patch size, the total amount of habitat, and the type, extent, and proximity of land-use activities, including development, agriculture, and roads, are important determinants of ecological integrity, affecting species richness, abundance, composition, and breeding success. The literature also suggests that a complex of upland and wetland habitats is needed to sustain wetland-dependent amphibian and reptile populations. Because the amount of suitable habitat is a dominant factor in determining species persistence, protecting relatively large tracts of high-quality habitat increases the probability of long-term maintenance of characteristic plant and animal species and communities.

An assessment that focuses on the specific habitat requirements of individual species or communities, represented by habitat patches, depends on the species selected for study. Some Pinelands species are found within a narrow range of habitats and others occur across the landscape. In effect, every natural habitat is probably important to some characteristic Pinelands species. A number of studies reviewed in Appendix 1 indicate that the extent of habitat is generally more important than spatial patterns and that focusing conservation efforts on the total amount of habitat in a landscape might be a more fruitful approach. This approach considers elements of the landscape as a heterogeneous functional mosaic instead of a collection of patches in a matrix (Forman 2002, Murphy and Lovett-Doust 2004).

Although a debate over the relative importance of small versus large ecological reserves, referred to as the SLOSS debate (Single Large or Several Small reserves), drew much attention in the 1970’s (Diamond 1975, Diamond et al., 1976, Simberloff and Abele 1976), many of the studies reviewed in Appendix 1 support Forman’s (1995) statement that no number of small patches can achieve the values of large patches. Important ecological values provided by large patches compared to small patches include water-quality protection, connectivity of low-order stream networks, habitat for interior-forest species, and the area needed to maintain natural-disturbance regimes (Forman 1995).

An ecosystem approach that places priority on large tracts of land does not discount the potential value of small reserves, which can contribute to overall biodiversity of a region and provide a means of protecting small populations of rare plants (Järvinen 1982, Simberloff and Gotelli 1984, Lesica and Allendorf 1992) and insects (Tscharntke et al. 2002). Lesica and Allendorf (1992) recognize the possible value of protecting small populations, with the caveat that small populations are not equivalent to larger ones for conservation purposes, are subject to loss of rare alleles, and are more prone to extinction from stochastic environmental events. Small reserves must be protected and managed more vigorously than large reserves (Noss 1983). Although small reserves are not a substitute for large ones, in some instances, small-habitat fragments might be the only remnants of a rare-habitat type, could harbor species not included in large reserves (Shafer 1995, McCoy and Mushinsky 1999), or might represent a broader spectrum of habitats and species diversity than larger reserves (Tscharntke et al. 2002). The value of a single, large plant reserve compared to several small ones depends on the species and habitat of interest (Järvinen 1982). Perhaps the best way to ensure the survival of rare-plant species or habitats types found in small-habitat fragments is to identify known locations and implement appropriate conservation strategies such as acquisition, regulation, or zoning. Fortunately, many rare Pinelands plant species are associated with wetlands, which are habitats that are afforded a relatively high level of protection in the region and throughout New Jersey (Breden et al. 2006).

It should be noted that some species might show a time-delayed response to fragmentation, habitat loss, and changes in the spatial configuration of habitat (Tilman et al. 1994, Loehle and Li 1996, Banks 1997, Findlay and Bourdages 2000, Fahrig 2001, Lindborg and Eriksson 2004). Following fragmentation, isolated populations of long-lived species may persist for long periods without successful reproduction or recruitment (Saunders et al. 1991). Thus, any landscape-integrity assessment may overestimate the actual long-term integrity of the landscape for some species or communities.

Evaluating landscape integrity. Because the manner in which a patch is characterized influences the outcome of a landscape analysis, it is necessary to describe patches with specific criteria that reflect the objective of the analysis (Turner et al. 2001). Determining the composition and spatial dimensions of a landscape patch is a matter of scale that depends both on the resolution and interpretation of the data used. The 2002 New Jersey Department of Environmental Protection land-use/land-cover data (NJDEP 2007), which uses a modified Anderson classification (Anderson et al. 1976), includes 40 Pinelands-habitat types, with greater detail given for upland forests than to wetlands. The resolution of Pinelands wetlands data increases when the Cowardin classification (Cowardin et al. 1979) is used. With modifying terms describing water chemistry, soil, and special conditions, the Cowardin classification yields 230 different wetland-cover types in the Pinelands compared to 23 based on the modified Anderson classification (NJDEP 2007). As an example, patches included in the modified Anderson-based Atlantic white cedar (*Chamaecyparis thyoides*) swamp class may be described by the Cowardin classification as an Atlantic white cedar swamp with or without a broad-leaved deciduous (e.g., red maple, *Acer rubrum*) or needle-leaved evergreen (e.g., pitch pine) component. Greater resolution along this hierarchy of scale may reveal many smaller patches of different cover types within a cedar swamp (Zampella and Lathrop 1997) or reflect variations in species composition (Laidig and Zampella 1999), tree structure (Gibson and Good 1986, Zampella et al. 1999), or within-patch microhabitat (Ehrenfeld 1995a, 1995b, Allison and Ehrenfeld 1999).

Rather than focus on individual habitats, such as Atlantic white cedar swamps, the landscape-integrity assessment was based on an evaluation of a single composite Pinelands-habitat type composed of Anderson-level upland-forest, wetland, and water patches, relative to the extent of non-habitat, represented by developed land, upland agriculture, wetland agriculture, managed wetlands, barren land, and roads (Table 2, Figure 4). Because the NJDEP land-use/land-cover data do not provide a complete depiction of roads, the paved surface of major roads delineated by the New Jersey Department of Transportation (NJDOT 2005) were buffered, delineated as polygons, and merged with the land-use/land-cover data (NJDEP 2000, 2007).[2] The habitat/non-habitat polygon (vector) data were rasterized, creating 10×10-m (32.8×32.8-ft) cells (pixels) that overlapped with the geographic extent of a digital-elevation model based on 10-meter digital-elevation grids (NJDEP 2002).

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[2] Road buffers were based on paved-road widths measured on-screen using 2002 digital aerial photographs. Measured road widths for New Jersey Department of Transportation route subtypes were as follows: U.S. Routes (44 ft); N.J. Routes (42 ft); Toll Authorities (46 ft); 500 Series County Routes (32 ft); 600 Series County Routes (30 ft); and Ramps (27 ft). The road widths for each road type were averages based on measurements made at fifty random points. Local roads were not buffered because nearly all of them were already included in non-habitat polygons. Conversion: 1 ft = 0.3048 m

Table 2. Pinelands land-use/land-cover habitat and non-habitat classification. The four-digit codes represent the NJDEP (2000, 2007) modified-Anderson land-use/land-cover categories. Agricultural wetlands (modified) are referred to as wetland agriculture throughout the report. A few land-use/land-cover types are not present in both the NJDEP 2000 and 2007 data set.

Pinelands habitat	Code	Non-habitat	Code
Upland forest		Wetlands	
Deciduous forest	4100	Managed wetland in maintained lawn greenspace	1750
Deciduous forest, 10-50% crown closure	4110	Managed wetland in built-up maintained rec. area	1850
Deciduous forest, >50% crown closure	4120	Agricultural wetlands (modified)	2140
Coniferous forest	4200	Managed wetlands (modified)	8000
Coniferous forest, 10-50% crown closure	4210	Barren land	
Coniferous forest, >50% crown closure	4220	Beaches	7100
Plantation	4230	Extractive mining	7300
Coniferous/deciduous forest	4310	Altered lands	7400
Mixed forest (>50% coniferous with 10-50% crown closure)	4311	Transitional areas	7500
Mixed forest (>50% coniferous with >50% crown closure)	4312	Undifferentiated barren lands	7600
Deciduous/coniferous forest	4320	Developed land	
Mixed forest (>50% deciduous with 10-50% crown closure)	4321	Residential	1100
Mixed forest (>50% deciduous with >50% crown closure)	4322	Residential, high density or multiple dwelling	1110
Brushland/shrubland	4400	Residential, single unit, medium density	1120
Old field (<25% brush covered)	4410	Residential, single unit, low density	1130
Phragmites dominated old field	4411	Residential, rural, single unit	1140
Deciduous brush/shrubland	4420	Mixed residential	1150
Coniferous brush/shrubland	4430	Commercial/services	1200
Mixed deciduous/coniferous brush/shrubland	4440	Military installations	1211
Severely burned upland vegetation	4500	Industrial	1300
Water		Transportation/communications/utilities	1400
Streams and canals	5100	Major roadway	1410
Natural lakes	5200	Bridge over water	1419
Artificial lakes	5300	Airport facilities	1440
Tidal rivers, inland bays and other tidal waters	5410	Upland rights-of-way developed	1462
Dredged lagoon	5420	Stormwater basin	1499
Wetlands		Industrial/commercial complexes	1500
Wetland rights-of-way (Modified)	1461	Mixed urban or built-up land	1600
Former agricultural wetland (becoming shrubby, not built-up)	2150	Other urban or built-up land	1700
Saline marshes	6110	Cemetery	1710
Saline marsh (low marsh)	6111	Cemetary on wetland	1711
Saline marsh (high marsh)	6112	Recreational land	1800
Freshwater tidal marshes	6120	Athletic fields (schools)	1804
Vegetated dune communities	6130	Stadium theaters, cultural centers, and zoos	1810
Phragmites dominated coastal wetlands	6141	Upland agriculture	
Deciduous wooded wetlands	6210	Cropland and pastureland	2100
Coniferous wooded wetlands	6220	Orchards/vineyards/nurseries/horticultural areas	2200
Atlantic white cedar swamp	6221	Confined feeding operations	2300
Deciduous scrub/shrub wetlands	6231	Other agriculture	2400
Coniferous scrub/shrub wetlands	6232		
Mixed scrub/shrub wetlands (deciduous dominated)	6233		
Mixed scrub/shrub wetlands (coniferous dominated)	6234		
Herbaceous wetlands	6240		
Phragmites dominated interior wetlands	6241		
Mixed forested wetlands (deciduous dominated)	6251		
Mixed forested wetlands (coniferous dominated)	6252		
Severely burned wetlands	6500		
Disturbed wetlands (modified)	7430		
Developed land			
Upland rights-of-way undeveloped[1]	1463		

[1] This land-use class was inadvertently placed in the non-habitat category in the initial release of the EIA report. It is now appropriately included in the habitat category.

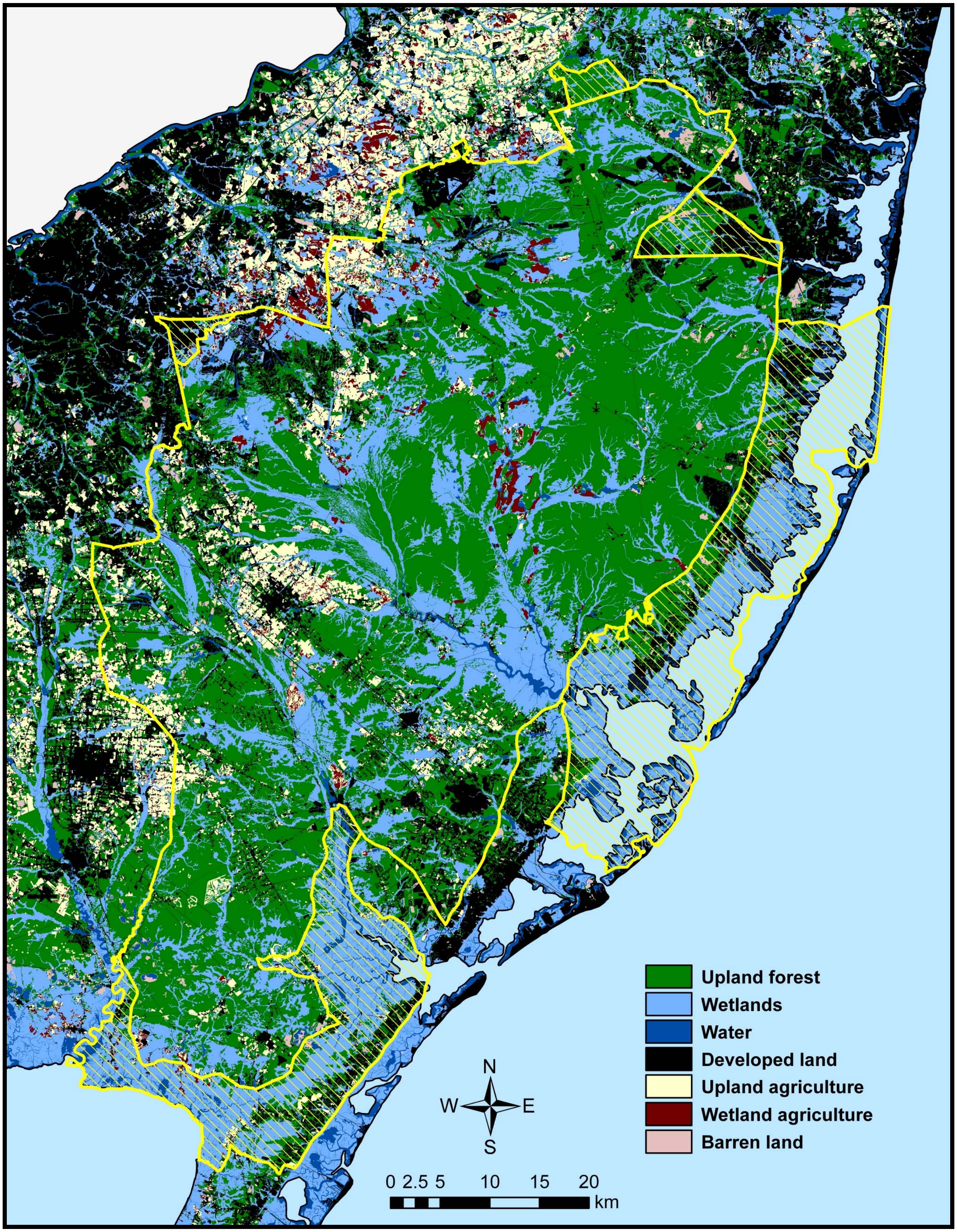

Figure 4. Pinelands Area land-use/land-cover in 2002 (NJDEP 2007). The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

Landscape integrity was characterized using a moving window to measure the amount of Pinelands habitat within a circle, described as an area of influence or neighborhood (Appendix 2). In a moving-window analysis, a "window" moves across a layer of rasterized or cell-based spatial data, performs a specified calculation on the data in the window, and assigns the result of that calculation to the center cell in the window (Figure 5). The window then moves to the next cell, performs the calculation again, and applies the results to the center cell of that neighborhood. This process continues until all the cells in the input-raster layer have been analyzed and an output-raster layer with the new values is created.

The landscape-integrity assessment included an analysis of separate 10×10-m Pinelands-habitat cells using a window with a 1,000-m (3,281-ft) radius.[3] The reasons for using a moving-window approach and selecting a window with a radius of 1,000 m are discussed in Appendix 2. The result of the analysis was a data layer composed of about 31 million Pinelands-habitat cells, with each cell assigned a landscape-integrity score represented by the percentage of habitat in the surrounding window. High landscape integrity was equated with a high percentage of surrounding Pinelands habitat. The results of the landscape-integrity analysis are presented in Appendix 4.

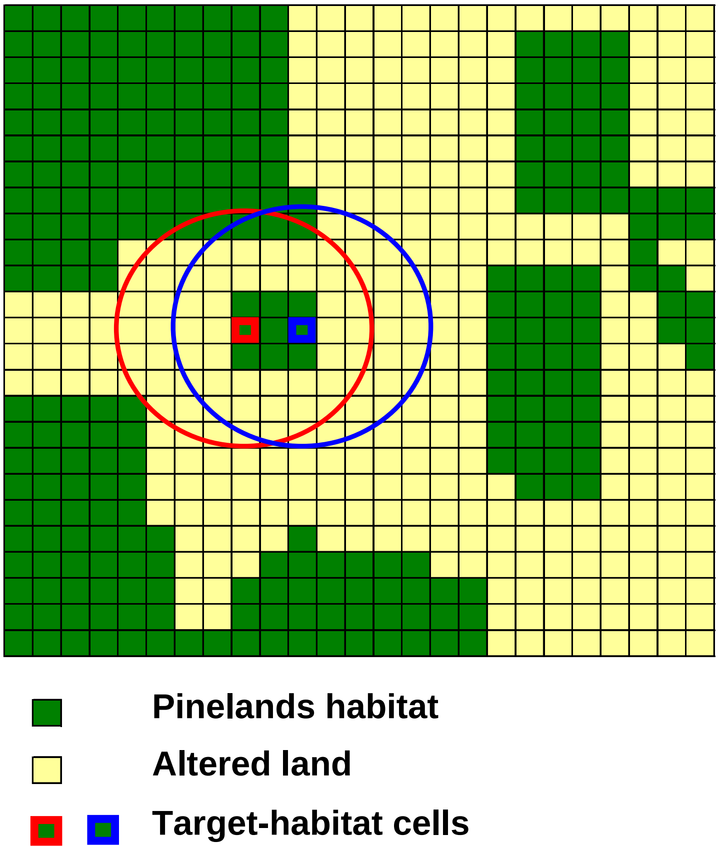

Figure 5. An example of a moving window. The window moves across a layer of rasterized or cell-based spatial data, performs a specified calculation on the data in the window, assigns the result of that calculation to the center cell in the window, and moves on to the next cell.

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[3] The Neighborhood Statistics tool in the ArcGIS version 9.2 Spatial Analyst extension (ESRI 2005) was used to measure the proportion of Pinelands habitat in the neighborhood of each cell.

Aquatic Integrity

Definition: Aquatic integrity is a measure of the percentage of land in a watershed that is neither developed land nor upland agriculture.

Background. Studies conducted in the Pinelands and throughout North America indicate that land use is a major factor influencing the status of aquatic systems (Appendix 1). Agriculture and urban activities are a significant source of nitrogen, phosphorus, and other dissolved solids found in surface waters and groundwaters. Stream communities are also affected by the extent of urban and agricultural lands in a watershed, although the effect of agriculture is more variable than that of urban land. In the Pinelands, water-quality degradation, characterized by increases in pH, specific conductance, and dissolved solids, is related to basin-wide, upland-land uses. Nonpoint sources are primarily responsible for degradation of Pinelands streams, but stormwater runoff may be an important source of some pollutants in more heavily developed areas. The composition of Pinelands aquatic communities also varies in relation to the percentage of developed land and upland agriculture in a watershed. A primary biological response to watershed disturbance in the Pinelands is the occurrence of diatoms characteristic of circumneutral waters and the presence of nonnative plants, fish, and frogs.

Land use can be related to stream conditions at different scales, including a local stream reach, an entire stream with an adjacent terrestrial buffer, or an entire watershed. North American stream studies assessing the effects of scale and the proximity of land use on water quality and biological communities have produced varying results, with some studies indicating that land use near streams has a greater effect than watershed-wide land use, others showing the opposite effect, and some indicating that differences between local and watershed-wide land-use effects are similar. These contrasting results may be due to differences in land-use patterns within a watershed.

Groundwater discharging from the Kirkwood-Cohansey aquifer is the dominant source of flow to Pinelands streams, with discharge to a point in a stream originating from sources that are near and far from the stream. Groundwater-travel times from recharge areas to a stream increase with distance to the stream, suggesting that land uses closer to a stream may have a greater influence on surface-water quality than distant land uses. However, available evidence indicates that including the proximity of developed land and upland agriculture to a stream does not improve the relationships between Pinelands surface-water quality and the proportion of watershed-wide land uses. Watershed-wide land use has also been shown to be a good indicator of the status of biological communities in Pinelands streams and impoundments.

Evaluating aquatic integrity. The aquatic-integrity assessment was based on the assumption that ecological integrity increases as the percentage of developed land and upland agriculture (altered land) in a watershed decreases. Because water quality and fish, anuran, plant, and diatom assemblages in stream basins with wetland agriculture are similar to streams in forested-basins, wetland agriculture was not included in the altered-land class. Neither managed wetlands nor barren land was included in the altered-land class because documentation indicating that these land-use types impact Pinelands water quality is lacking.

Watersheds were created using NJDEP (1996) stream data, a digital-elevation model (NJDEP 2002), and Arc Hydro (ESRI 2007). Arc Hydro is a water resources tool designed to work within ArcGIS (ESRI 1999-2006). Points, referred to as pour points, were placed immediately above the confluence of streams or stream segments and the DEM was used to delineate the total area contributing stream flow to each pour point. To reduce variations in scale, streams or segments of streams that were two or more times as long as the median length of all first-order Pinelands streams were divided until they were less than twice the length of the median value. The total upstream area draining to a pour point represents the watershed. The term drainage unit refers to the portion of the watershed between pour points (Figure 6). The watershed-delineation process produced a total of 5,328 Pinelands Area drainage units and associated watersheds. The boundary of each drainage unit was checked against U. S. Geological Survey 1:24,000-scale topographic maps.

Aquatic integrity, defined as the percentage of a watershed that was neither developed land nor upland agriculture (Table 2), was determined for each watershed and assigned to the associated stream segment and drainage unit defined by the lower and upper pour points for that segment. The integrity score assigned to a drainage unit was then assigned to every 10×10-m cell in that drainage unit. The results of the aquatic-integrity assessment are presented in Appendix 4.

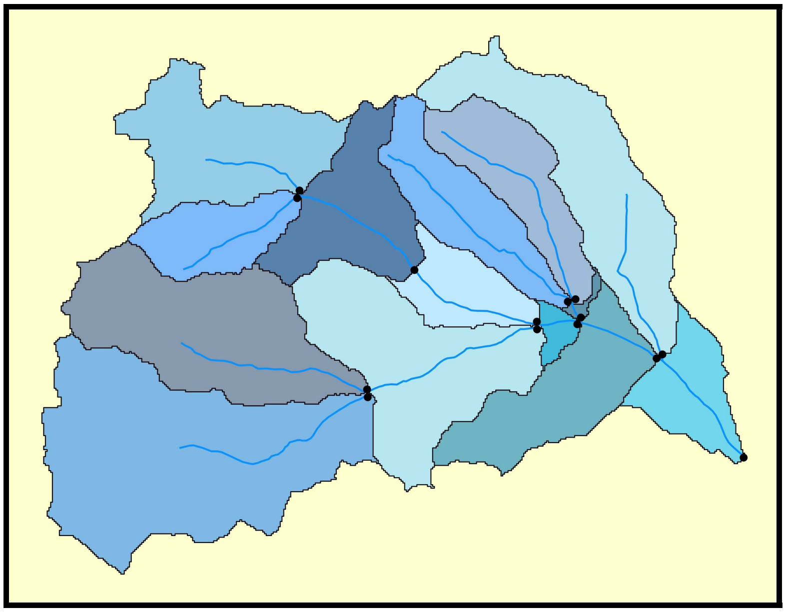

Figure 6. A watershed is the total upstream area draining to a pour point. Pour points are shown as dots. The term drainage unit refers to the portion of a watershed between pour points. Fourteen separate drainage units are depicted here.

Wetland-drainage Integrity

Definitions: Wetland-drainage integrity is a measure of the percentage of land in a wetland-drainage unit that is neither developed land nor upland agriculture. A wetland-drainage unit is a discrete area of wetlands and the adjacent uplands that contribute surface water and groundwater to the wetlands.

Background. The review of aquatic and wetland literature presented in Appendix 1 indicates that land uses in adjacent uplands influence both the quantity and quality of groundwater flowing to wetlands, which in turn can affect the composition of plant and animal communities associated with wetlands. Available evidence suggests that the proximity of developed land and upland agriculture does not substantially improve the relationship between land use and Pinelands surface-water quality. Similar information is lacking for Pinelands wetlands, and the question of whether land uses closer to a wetland have a greater influence on wetland water quality and hydrology than distant land uses remains unanswered. However, an analysis of the potential effect of land-use proximity on the hydrologic integrity of wetlands indicated that existing land-use patterns in the Pinelands are such that when comparing the relative value of wetlands based on the extent of developed and upland-agricultural land located upgradient from the wetlands, the relationship was similar regardless of whether near, far, or total land use in a wetland-drainage unit was considered (Appendix 3).

Evaluating wetland-drainage integrity. To determine the extent of adjacent uplands that might affect the quantity and quality of groundwater flowing to wetlands, the drainage units developed for the aquatic-integrity assessment were split along streams to create wetland-drainage units with the assumption that the uplands in these drainage units contribute flow to the adjacent section of wetlands (Figure 7). By placing pour points at the upper limit of mapped streams, additional units were delineated for a total of 12,516 individual Pinelands Area wetland-drainage units.

Wetland-drainage integrity, defined as the percentage of a wetland-drainage unit that was neither developed land nor upland agriculture (Table 2), was determined for each wetland-drainage unit. Because including only uplands would exaggerate the possible influence of a high percentage of developed land and upland agriculture (altered land) in wetland-drainage units with a low percentage of uplands, the percentage of altered land in the entire wetland-drainage unit was selected as the measure of wetland integrity. The integrity score assigned to a wetland-drainage unit was assigned to every 10×10 m cell in that unit. The results of the wetland-drainage-integrity assessment are presented in Appendix 4.

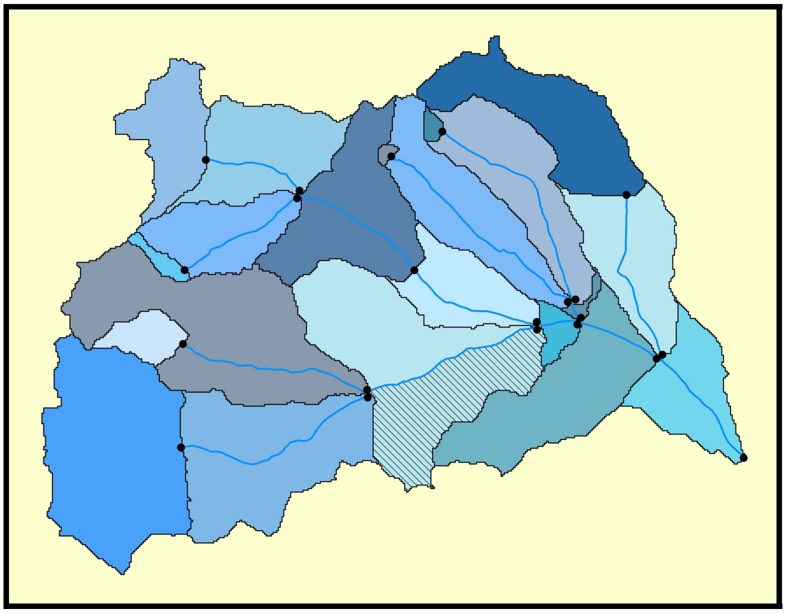

Figure 7. Wetland-drainage units were created by splitting drainage units along streams. Additional units were delineated by placing a pour point at the upper limit of mapped streams. The hatched area represents an example of a wetland-drainage unit.

PINELANDS ECOLOGICAL INTEGRITY

A Pinelands ecological-integrity score was determined for every 10×10-m Pinelands-habitat cell in the Pinelands Area raster layer. This score represents an average of the landscape-, aquatic-, and wetland-drainage-integrity scores. The results of the ecological-integrity assessment, which was based on 2002 land-use/land-cover data, are summarized spatially in Figure 8.

Pinelands habitat and non-habitat covered 82% and 18% of the Pinelands Area, respectively (Figure 9). Fifty-one percent of the Pinelands Area fell within the highest Pinelands ecological-integrity class (90.1-100%). This class made up 62% of all Pinelands habitat. Less than three percent of the area displayed scores less than or equal to 50%.

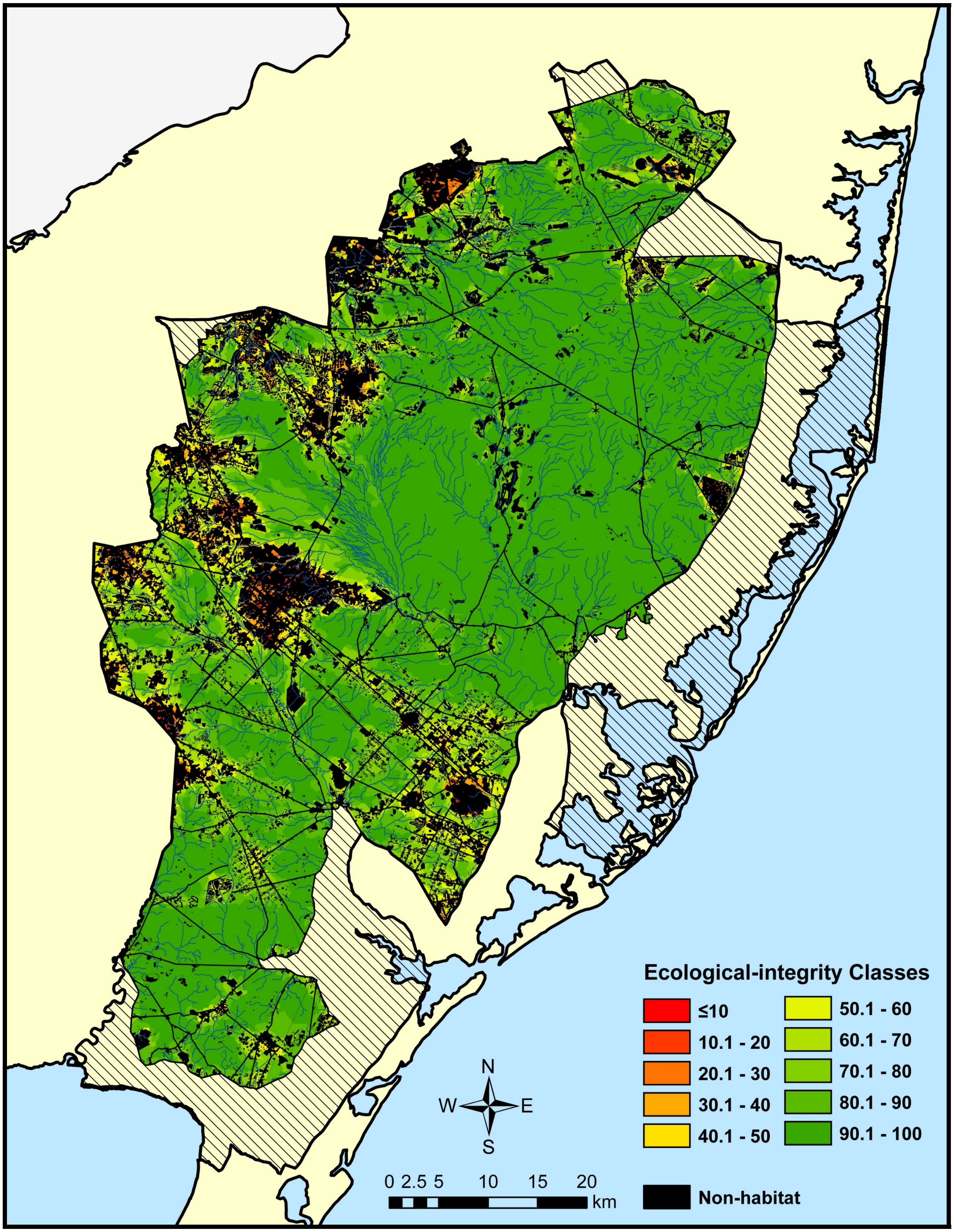

Figure 8. Ecological integrity of Pinelands habitat. Ecological-integrity for each habitat cell is an average of the landscape-, aquatic-, and wetland-drainage-integrity scores based on 2002 land-use/land-cover data. The 90.1-100% class represents the highest level of ecological integrity. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

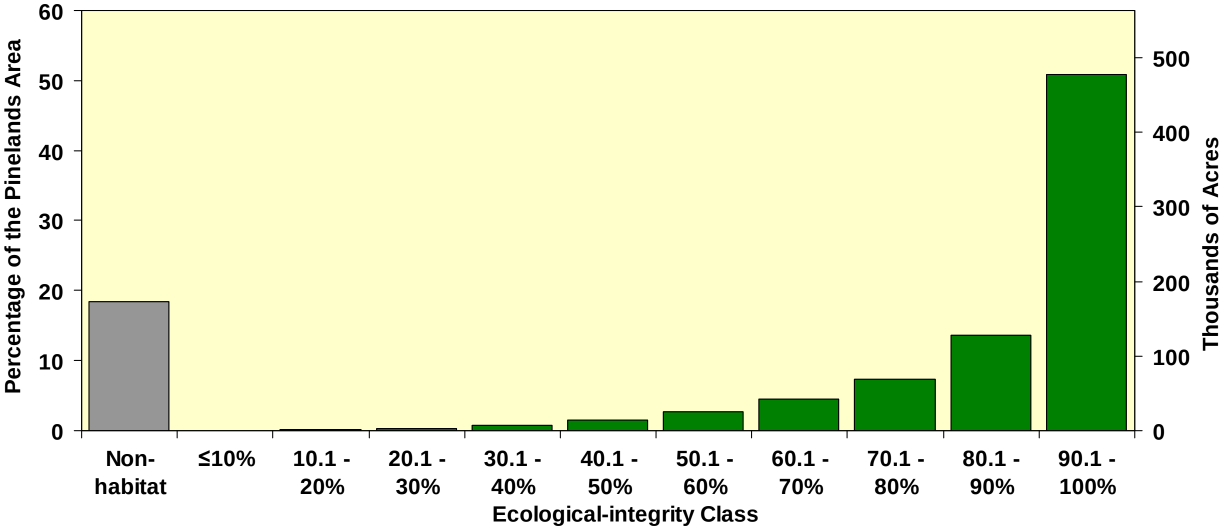

Figure 9. The percentage of the Pinelands Area within each of ten ecological-integrity classes ranging from ≤10% (lowest integrity) to 90.1-100% (highest integrity). Ecological integrity represents an average of landscape-, aquatic-, and wetland-drainage-integrity scores assigned to every Pinelands-habitat cell. Non-habitat includes developed land, upland agriculture, wetland agriculture, managed wetlands, barren land, and roads.

Relationship among Integrity Measures

Spearman rank correlation was used to quantify the relationship among the different integrity measures based on a sample of 300 random-habitat cells. The basis for selecting the 300 random-cells is discussed in Appendix 2. As shown in Table 3, the correlations between the 2002 Pinelands ecological-integrity scores and the landscape-, aquatic-, and wetland-drainage-integrity scores were strong. The relationships among the individual integrity assessments were weaker, which suggests that each assessment described a different aspect of ecological integrity.

To further evaluate the relationship among the different integrity measures, a cell-by-cell comparison of the raster layer depicting the ecological integrity of Pinelands habitat in 2002 and those depicting landscape-, aquatic-, and wetland-drainage-integrity for the same period was summarized using contingency tables (Table 4). In each case the majority of cells within a particular Pinelands ecological-integrity class fell within the same class in the landscape-, aquatic-, and wetland-integrity assessments. For example, 69% of the cells within the 10.1-20% Pinelands ecological-integrity class fell within the 10.1-20% landscape-integrity class. These relationships were most consistent for cells in the 90.1-100% classes. Most differences between the Pinelands ecological-integrity scores and the other integrity measures involved cells that fell within the next lowest or highest landscape-, aquatic-, or wetland-drainage-integrity class.

Table 3. Spearman rank correlations between Pinelands ecological-, landscape-, aquatic-, and wetland-drainage-integrity scores for 300 random-habitat cells based on 2002 land-use/land-cover data. All correlations are significant at $p < 0.001$.

	Ecological	Landscape	Aquatic	Wetland-drainage
Ecological integrity	1.00	-	-	-
Landscape integrity	0.92	1.00	-	-
Aquatic integrity	0.93	0.75	1.00	-
Wetland-drainage integrity	0.89	0.81	0.77	1.00

Table 4. Contingency tables showing the cell-to-cell relationships between composite Pinelands ecological-integrity scores and landscape-, aquatic-, and wetland-drainage-integrity scores given as a percentage of the Pinelands-habitat cells in each ecological-integrity class. In each case the majority of habitat cells within a particular Pinelands ecological-integrity class fell within the same class in the landscape-, aquatic-, and wetland-integrity assessments. For example, 69% of the cells within the 10.1-20% Pinelands ecological-integrity class fell within the 10.1-20% landscape-integrity class. The table was based on the integrity scores for all Pinelands-habitat cells obtained using 2002 land-use/land-cover data. Integrity classes are percentages (e.g., 10.1-20%).

		Landscape-integrity class									
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100
Ecological-integrity class	≤10	31	67	2	-	-	-	-	-	-	-
	10.1-20	8	69	19	4	1	-	-	-	-	-
	20.1-30	2	16	44	25	11	3	<1	-	-	-
	30.1-40	<1	5	23	35	27	7	2	<1	-	-
	40.1-50	-	1	5	20	39	25	8	2	<1	-
	50.1-60	-	<1	1	4	22	41	26	6	1	<1
	60.1-70	-	<1	<1	1	5	21	44	24	4	<1
	70.1-80	-	-	<1	<1	1	5	21	41	26	6
	80.1-90	-	-	-	-	<1	1	6	19	43	31
	90.1-100	-	-	-	-	-	-	-	1	9	89

		Aquatic-integrity class									
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100
Ecological-integrity class	≤10	76	24	-	-	-	-	-	-	-	-
	10.1-20	7	64	29	<1	-	-	-	-	-	-
	20.1-30	2	22	50	20	5	1	-	-	-	-
	30.1-40	<1	4	20	44	25	6	2	1	<1	-
	40.1-50	<1	1	5	17	37	31	7	2	1	<1
	50.1-60	-	<1	1	9	21	44	19	4	2	<1
	60.1-70	-	<1	1	2	9	23	39	21	5	1
	70.1-80	-	-	<1	2	4	11	23	34	20	5
	80.1-90	-	-	-	-	1	5	13	23	37	21
	90.1-100	-	-	-	-	-	-	-	3	11	86

		Wetland-drainage-integrity class									
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100
Ecological-integrity class	≤10	93	7	-	-	-	-	-	-	-	-
	10.1-20	20	47	30	3	-	-	-	-	-	-
	20.1-30	4	25	55	15	2	-	-	-	-	-
	30.1-40	<1	10	21	45	19	3	1	<1	-	-
	40.1-50	<1	1	6	29	37	19	6	1	<1	<1
	50.1-60	<1	<1	1	5	20	36	28	7	2	1
	60.1-70	-	<1	<1	1	5	16	42	22	9	6
	70.1-80	-	-	<1	<1	<1	3	14	34	29	20
	80.1-90	-	-	-	-	-	<1	1	9	35	56
	90.1-100	-	-	-	-	-	-	-	<1	2	98

Comparison of Pinelands Management Areas

A weighted-average method was used to rank the overall ecological integrity of Pinelands habitat in each Pinelands management area. For each of the nine management areas, the percentage of Pinelands habitat in each ecological-integrity class was multiplied by the upper range of the class (e.g., the weight for the 10.1-20% class equaled 20), the weighted percentages were summed and divided by 100, and the resulting management-area weighted-scores were ranked.

Pinelands Towns, Agricultural Production Areas, and Regional Growth Areas displayed the lowest overall ecological integrity, whereas the Preservation Area District, Special Agricultural Production Areas, and Forest Areas displayed the highest ecological integrity (Figures 10 and 11). The percentage of habitat within the five lowest ecological-integrity classes generally decreased along this Pinelands Town to Preservation Area gradient, and the percentage of habitat within the five highest ecological-integrity classes generally increased.

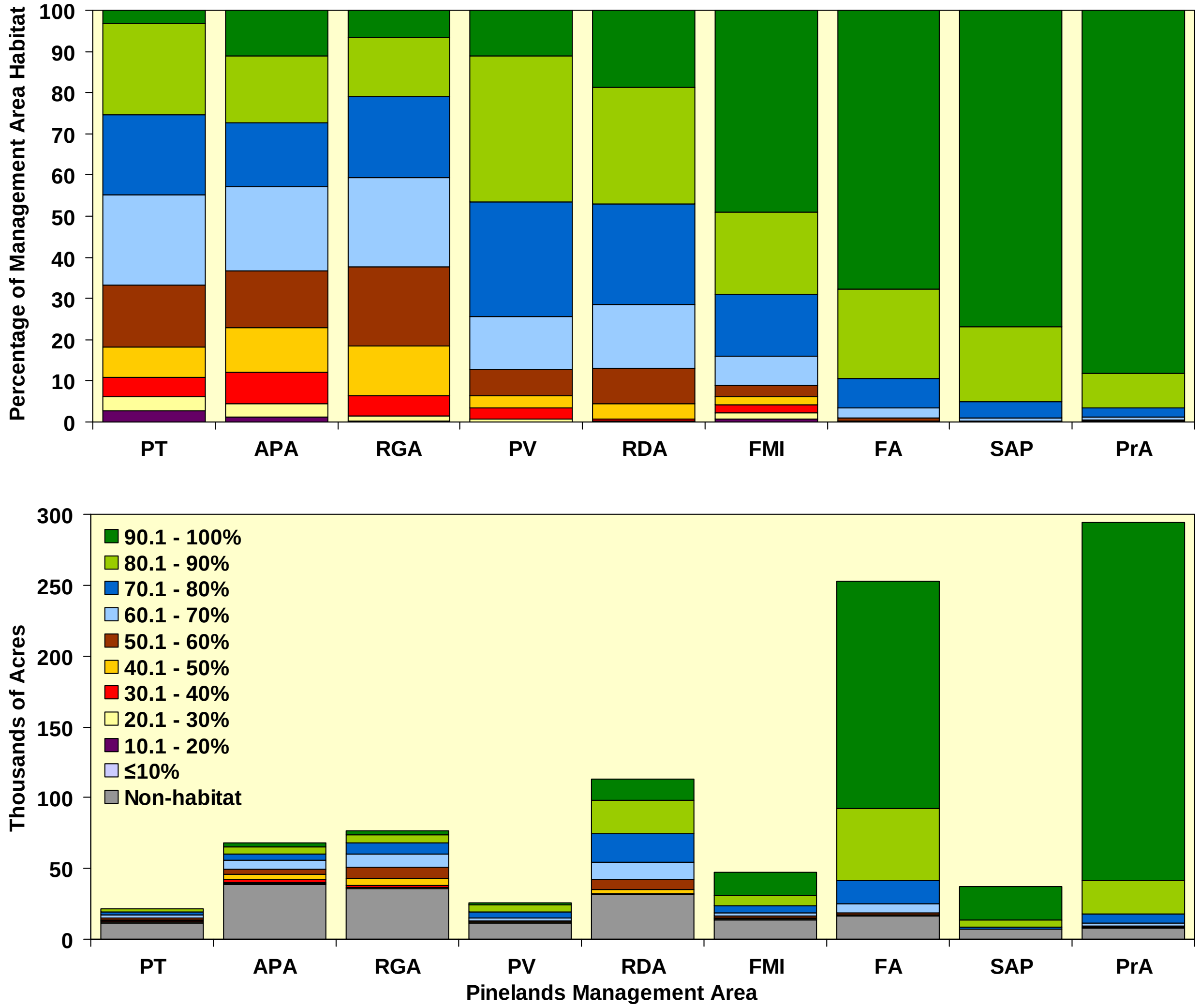

Figure 10. Ecological-integrity-class composition of Pinelands habitat and the acres of Pinelands habitat and non-habitat in each class in each Pinelands management area. Refer to Table 1 for Pinelands management-area codes.

Most areas of high integrity were found within the Preservation Area District, Special Agricultural Production Areas, and Forest Areas (Figure 10 and 11). These three management areas, which make up nearly two-thirds of the Pinelands Area, accounted for 92% of the habitat within the 90.1-100% ecological-integrity class and 63% of the habitat in the 80.1-90% class. Five percent of the land in these three management areas was classified as non-habitat. In contrast, Pinelands Towns, Agricultural Production Areas, and Regional Growth Areas accounted for 66% of the habitat with ecological-integrity scores ≤50%. These three management areas made up about 18% of the Pinelands Area, but accounted for 50% of the region's non-habitat. Fifty-two percent of the land area included in these three management areas was non-habitat.

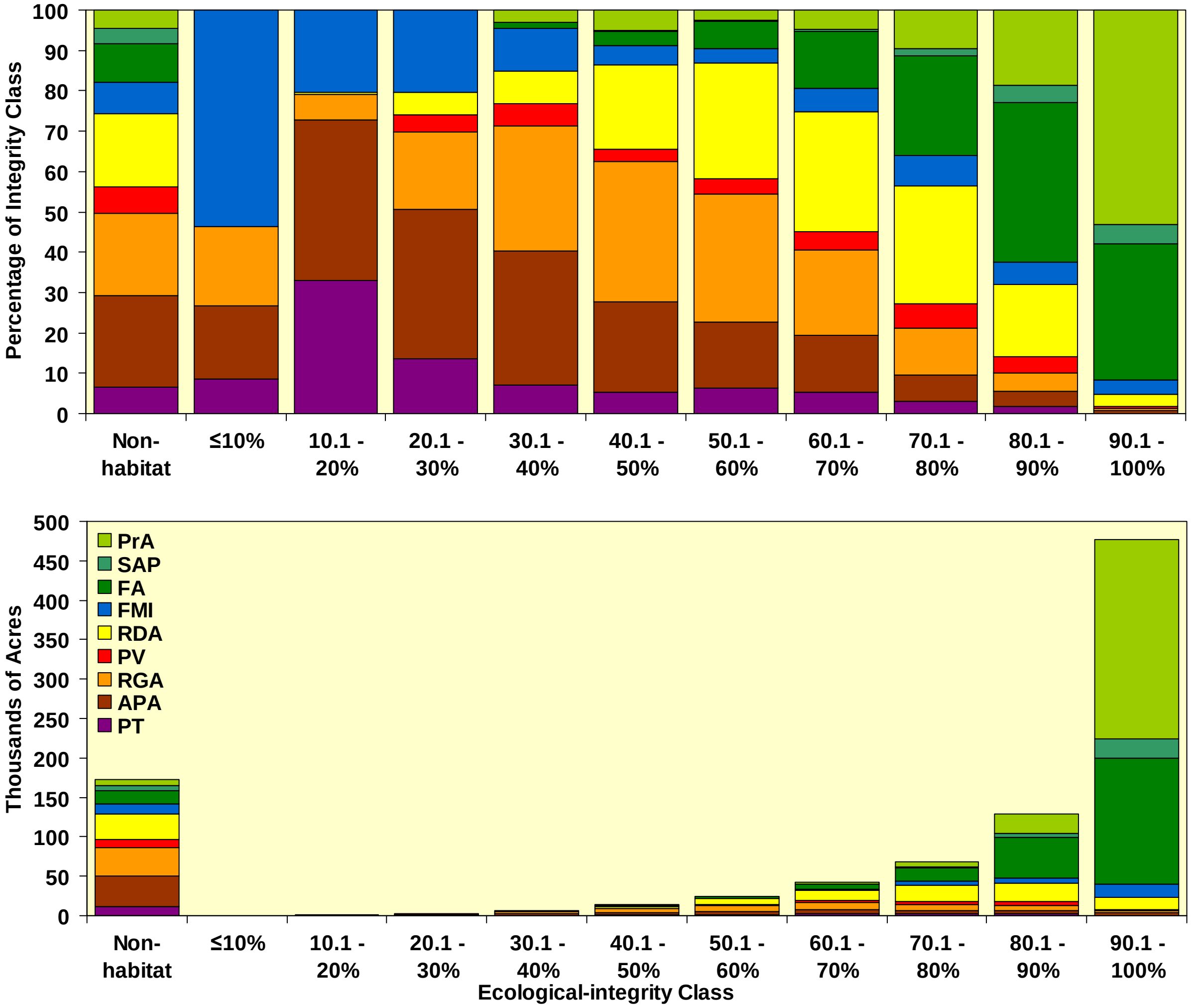

Figure 11. Pinelands management-area composition of ecological-integrity classes and the acres of Pinelands habitat in each class. Refer to Table 1 for Pinelands management-area codes.

Ecological Integrity by Municipality

Regional Growth Areas, Pinelands Towns, and Pinelands Villages. Because development is generally directed to Regional Growth Areas, Pinelands Towns, and Pinelands Villages, the ecological integrity of these development areas in each Pinelands municipality was analyzed with the same approach used in the previous management-area analysis. The same analysis was completed for Rural Development Areas.

The overall ecological integrity of development areas (Regional Growth Areas, Pinelands Towns, and Pinelands Villages) in each municipality was ranked using the weighted-average method (Figure 12). Development areas in municipalities such as Buena Borough, Wrightstown, Medford Lakes, Hammonton, and Berlin Borough displayed relatively low overall ecological integrity. At the opposite end of the ecological-integrity gradient, development areas in Maurice River Township, Little Egg Harbor Township, Lacey Township, Woodland Township, and Washington Township displayed the highest level of ecological integrity. However, to accurately assess the potential regional impact of developing lands of varying ecological-integrity, the actual areas that might be affected must be considered (Figures 13 and 14).

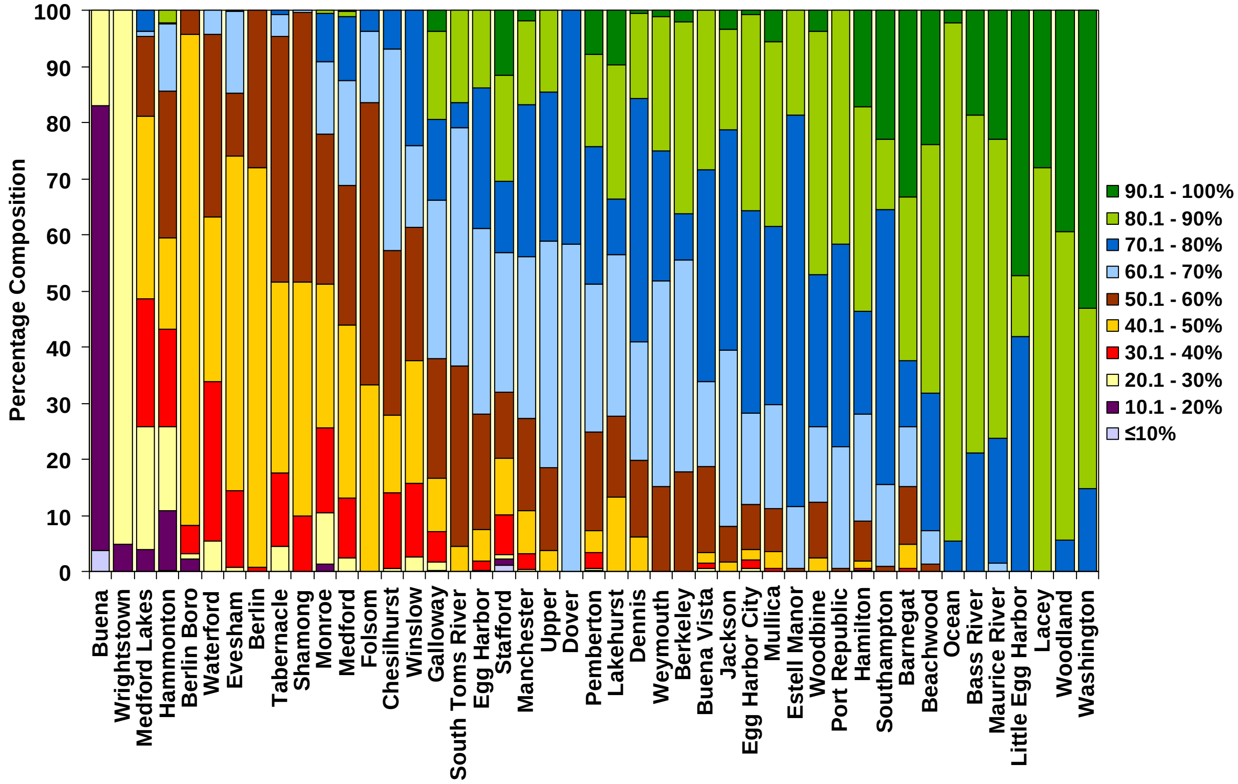

Figure 12. Ecological-integrity-class composition of habitat in Regional Growth Areas, Pinelands Towns, and Pinelands Villages by municipality. Municipalities are ordered by increasing ecological integrity.

In Figure 14, municipalities are ordered based on weighted-ecological-integrity scores and acres of upland habitat. Both upland acreage and overall ecological integrity is low in municipalities shown in the lower left corner of the figure. Development areas in municipalities shown in the upper right corner of the figure are characterized by greater upland acreage and higher ecological integrity.

Rural Development Areas. Rural Development Areas in municipalities such as Medford Lakes, Berlin Township, Plumsted Township, Shamong Township, and Waterford Township displayed relatively low overall ecological integrity, whereas Buena Vista Township, Hamilton Township, Upper Township, Maurice River Township, and Bass River Township displayed the highest level of ecological integrity (Figure 15). The acres of Pinelands habitat in each municipality are shown in Figures 16 and 17. Both upland acreage and overall ecological integrity was low in municipalities shown in the lower left corner of the Figure 17. Rural Development Areas in municipalities shown in the upper right corner of Figure 17 were characterized by greater upland acreage and higher ecological integrity.

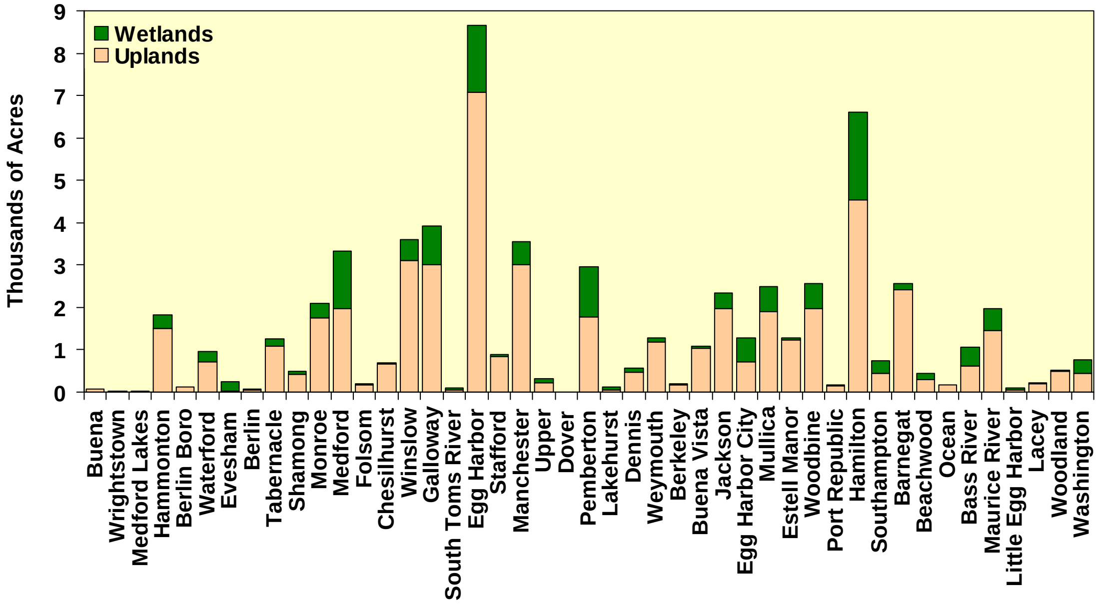

Figure 13. Acres of Pinelands habitat in Regional Growth Areas, Pinelands Towns, and Pinelands Villages. Municipalities are ordered by increasing ecological integrity.

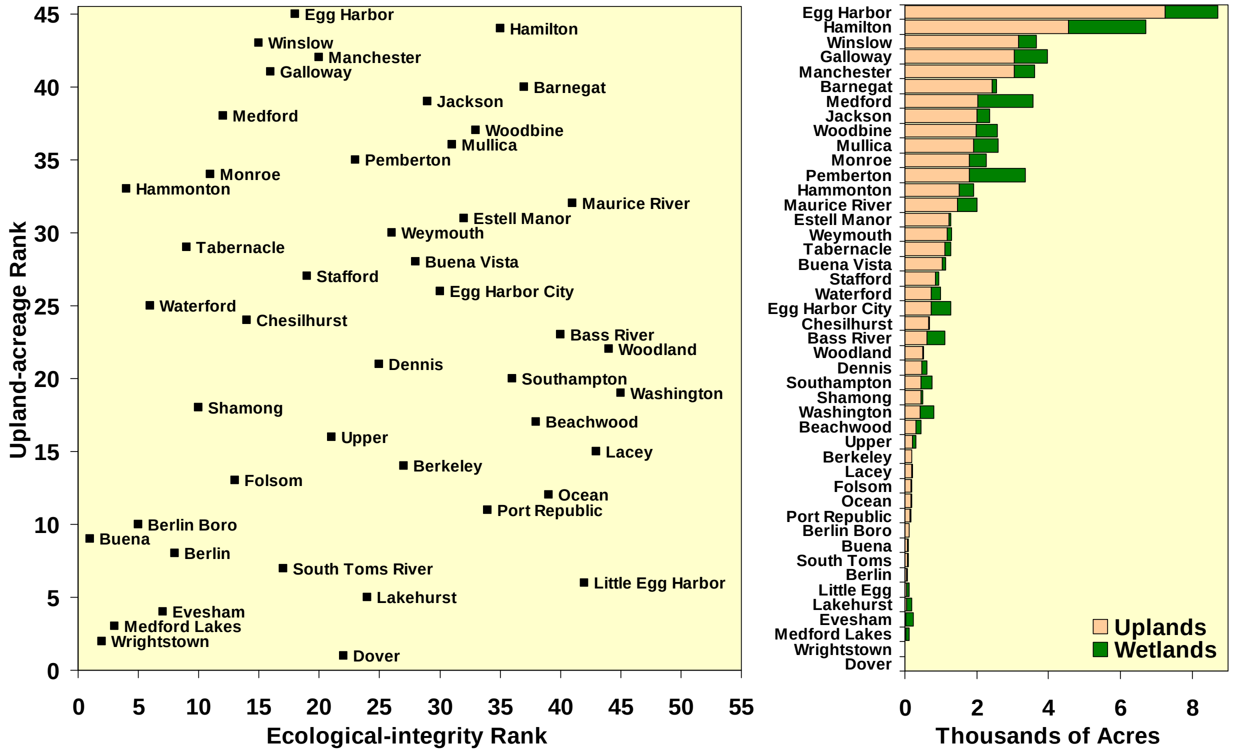

Figure 14. Municipalities ordered by increasing ecological-integrity rank and acres of upland habitat in Regional Growth Areas, Pinelands Towns, and Pinelands Villages.

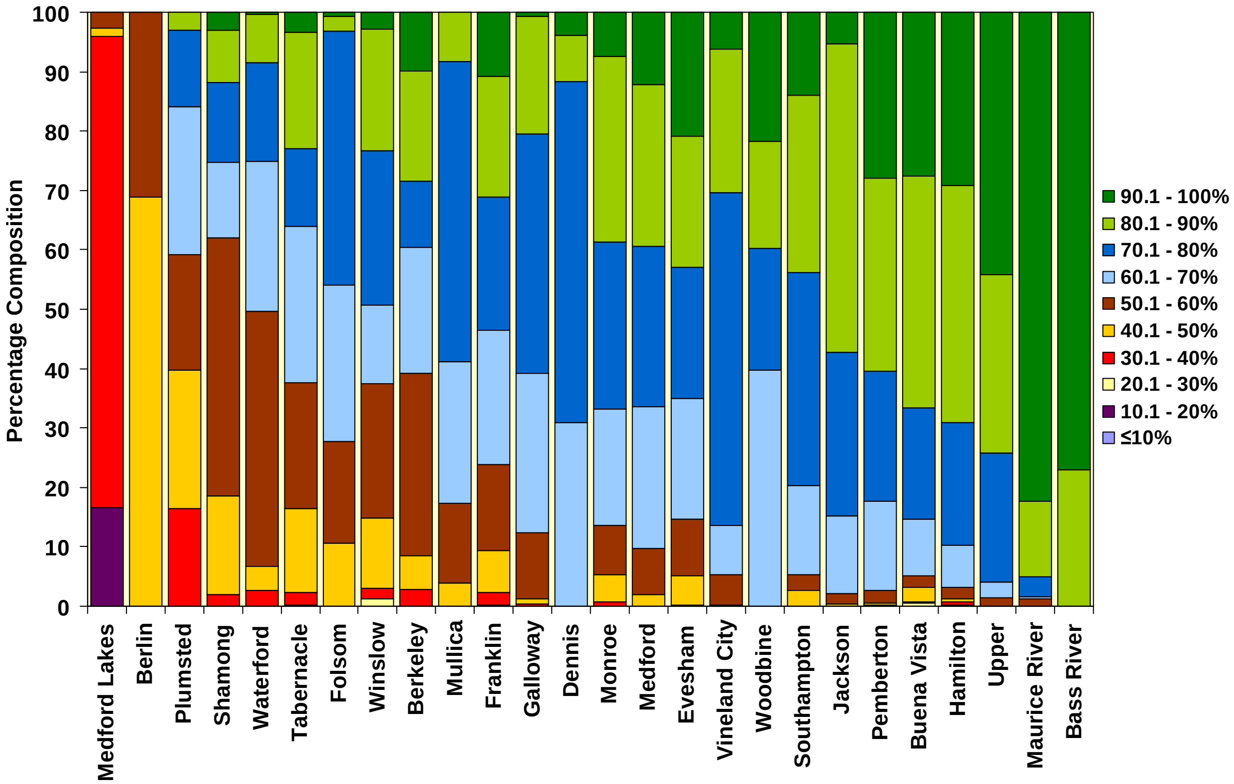

Figure 15. Ecological-integrity-class composition and acres of Pinelands habitat in Rural Development Areas by municipality. Municipalities are ordered by increasing ecological integrity.

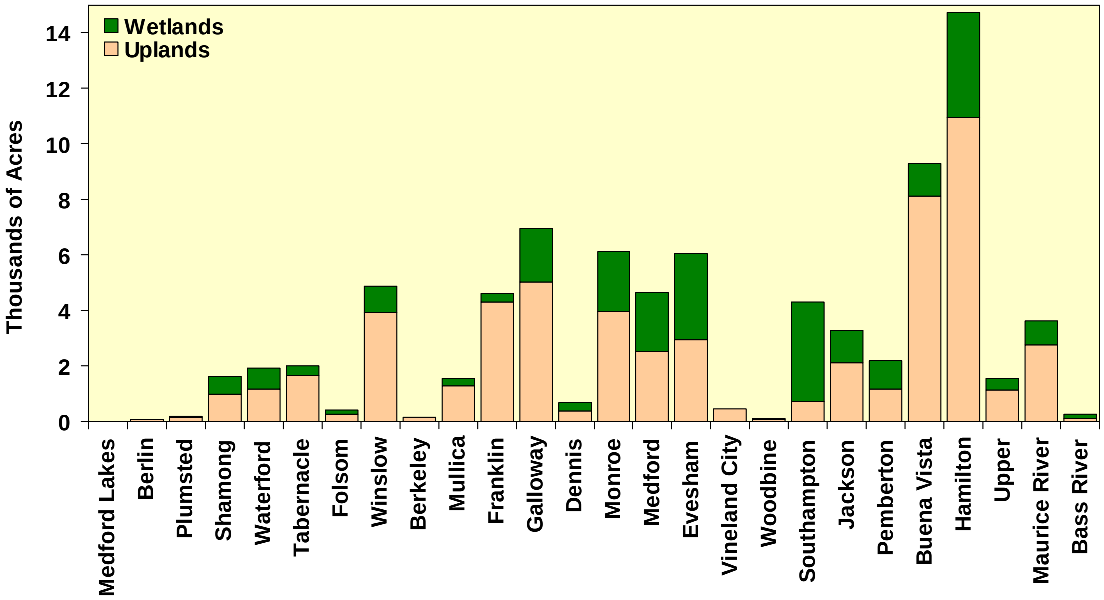

Figure 16. Ecological-integrity-class composition and acres of Pinelands habitat in Rural Development Areas by municipality. Municipalities are ordered by increasing ecological integrity.

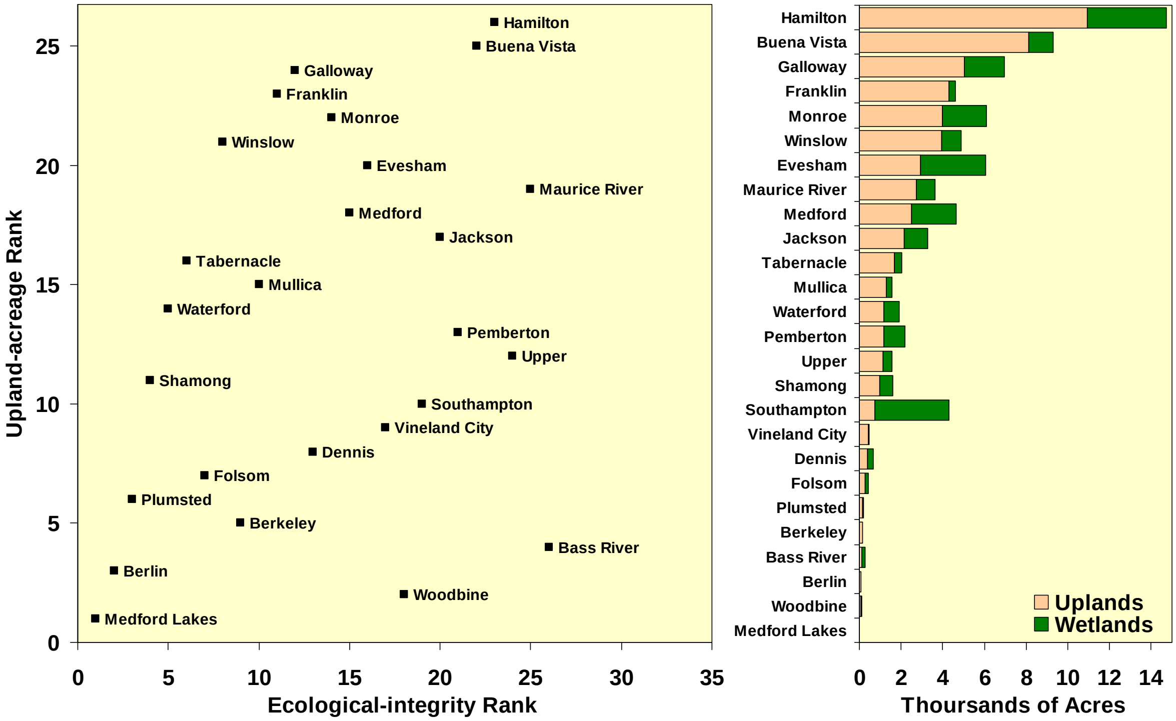

Figure 17. Municipalities ordered by increasing ecological-integrity rank and acres of upland habitat in Rural Development Areas.

Ecological Integrity, Habitat Diversity, and Rare Species Diversity

Pinelands ecological integrity as applied in this assessment was related to the extent and composition of unaltered landscapes and watersheds. Considering the possibility that small habitat fragments may be the only remnants of a rare habitat type, harbor species not included in large reserves, or might represent a broader spectrum of habitats and species diversity, the habitat types and rare-species diversity of lands within each ecological-integrity class were evaluated.

The percentage of each Anderson-based upland and Cowardin-based wetland cover type (NJDEP 2007) falling within each ecological-integrity class was summarized (Tables 5 and 6). All the major upland and wetland habitat types were well represented in the 90.1-100% ecological-integrity class. With two exceptions, at least 50% of the total acreage of all upland- and wetland-habitat types was found in the two highest classes. The exceptions included *Phragmites*-dominated old fields (2 acres) and deciduous-forest/emergent wetlands (7 acres).

The New Jersey Department of Environmental Protection, Office of Natural Lands Management (ONLM) priority sites[4] represent critically important areas of biological diversity, particularly areas with rare-plant species and ecological communities, with the caveat that the priority sites do not include all the known habitat for endangered and threatened species. Priority sites are categorized as either a macrosite or a standard site. Standard sites are usually smaller in size (less than 3,200 acres) than macrosites and can be found within the boundaries of a macrosite. The ONLM macrosite and standard-site data were rasterized and the percentage area of each site type was determined for the ten Pinelands ecological-integrity classes. Eighty-nine percent of the area designated as macrosites was within the 90.1-100% ecological-integrity class (Figure 18). Similarly, 82% of the area associated with standard sites fell within the highest ecological-integrity class.

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[4] Natural Heritage Program Priority Sites: http://www.nj.gov/dep/gis/crossaccept.htm#prisite_ca.

Table 5. Percentage of upland-forest-type area by ecological-integrity class. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

	Ecological-integrity class											
Upland-forest type	≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	Total acres	Total ha
Deciduous forest (10-50% crown closure)	<0.1	1.2	3.1	4.8	7.2	7.5	9.5	12.6	16.8	37.3	7,531	3,049
Deciduous forest (>50% crown closure)	<0.1	0.2	0.7	2.5	5.8	9.0	13.1	16.1	20.4	32.0	64,732	26,207
Coniferous forest (10-50% crown closure)	<0.1	<0.1	<0.1	0.2	0.4	0.7	1.6	3.2	8.1	85.7	41,253	16,702
Coniferous forest (>50% crown closure)	<0.1	<0.1	0.1	0.3	0.7	1.8	3.4	6.0	13.0	74.8	177,172	71,729
Plantation	-	0.6	3.8	0.6	4.3	7.5	11.4	13.6	28.8	29.4	817	331
Mixed forest (>50% coniferous with 10-50% crown closure)	-	0.2	0.7	0.6	2.1	3.2	6.7	11.8	15.0	59.7	6,033	2,442
Mixed forest (>50% coniferous with >50% crown closure)	<0.1	0.1	0.2	0.6	1.4	3.6	6.1	11.2	19.5	57.4	72,174	29,220
Mixed forest (>50% deciduous with 10-50% crown closure)	<0.1	0.1	0.4	1.0	2.4	3.6	6.8	12.2	17.0	56.4	7,043	2,852
Mixed forest (>50% deciduous with >50% crown closure)	<0.1	<0.1	0.2	0.7	1.5	3.7	7.8	11.2	20.7	54.3	75,478	30,558
Old field (< 25% brush covered)	0.1	1.3	1.8	2.8	5.9	6.2	7.8	10.7	16.6	46.9	7,577	3,067
Phragmites-dominated old field	-	-	-	100	-	-	-	-	-	-	2	1
Deciduous brush/shrubland	0.3	1.3	2.7	4.0	4.7	5.1	6.1	6.1	11.5	58.1	3,890	1,575
Coniferous brush/shrubland	<0.1	0.1	0.1	0.3	0.6	1.6	1.9	3.6	6.0	85.8	25,082	10,155
Severe-burned-upland vegetation	0.1	0.3	1.0	1.3	2.8	3.7	6.4	6.3	12.7	65.4	14,937	6,047
Upland forest-type richness	11	13	13	14	13	13	13	13	13	13	14	14

Table 6. Percentage of wetland-type area by ecological-integrity class. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

			Ecological-integrity class											
Wetland type	Cowardin code	Non-habitat[1]	≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	Total acres	Total ha
Estuarine-intertidal emergent	E2EM1	-	-	-	-	-	-	-	-	-	13.9	86.1	212	86
Estuarine-intertidal scrub-shrub	E2SS1	-	-	-	-	-	-	-	-	45.6	54.4	-	1	1
Lakes	L2OW	3.8	-	<0.1	0.1	1.0	4.2	8.0	8.3	10.0	23.9	40.8	5,516	2,233
Emergent (persistent vegetation)	PEM1	3.4	-	0.1	3.1	2.6	2.7	2.2	2.8	6.2	11.1	65.8	9,547	3,865
Emergent/deciduous scrub-shrub	PEM1/PSS1	4.7	-	0.4	0.5	0.6	3.1	1.3	3.1	8.9	13.5	64.0	2,157	873
Emergent/white-cedar and deciduous scrub-shrub	PEM1/PSS8/1	-	-	-	-	-	-	-	-	-	-	100	3	1
Emergent (non-persistent vegetation)	PEM2	-	-	-	-	-	-	-	-	-	-	100	6	2
Deciduous forest (e.g., hardwood swamps)[2]	PFO1	0.6	<0.1	0.3	0.9	2.0	4.1	5.4	8.4	14.3	24.0	40.1	53,604	21,702
Deciduous/broad-leaved-evergreen forest	PFO1/3	<0.1	-	-	-	-	0.3	3.8	8.8	9.4	21.4	56.4	1,680	680
Deciduous/needle-leaved-evergreen forest	PFO1/4	0.3	<0.1	<0.1	0.1	0.6	1.4	3.2	5.8	10.3	18.3	60.1	13,670	5,535
Deciduous/white-cedar forest	PFO1/8	0.1	-	-	-	0.2	0.5	1.4	3.4	6.8	25.5	62.2	4,510	1,826
Deciduous forest/emergent (persistent vegetation)	PFO1/PEM1	-	-	-	26.6	73.4	-	-	-	-	-	-	7	3
Deciduous forest/scrub-shrub[2]	PFO1/PSS1	0.2	-	0.6	0.1	2.4	1.8	1.4	5.3	5.2	22.7	60.2	1,453	588
Broad-leaved-evergreen forest	PFO3	-	-	-	-	-	9.5	0.5	-	2.0	34.5	53.5	27	11
Broad-leaved-evergreen/deciduous forest	PFO3/1	-	-	-	-	-	-	-	-	5.5	36.5	58.0	192	78
Broad-leaved-/needle-leaved evergreen forest	PFO3/4	-	-	-	-	-	-	-	-	-	12.2	87.8	105	43
Broad-leaved-evergreen/white-cedar forest	PFO3/8	-	-	-	-	-	-	-	-	-	1.3	98.7	14	6
Needle-leaved-evergreen forest (e.g. , pitch pine lowland)	PFO4	0.4	-	<0.1	<0.1	0.2	0.6	2.0	4.6	9.2	19.4	63.7	37,635	15,237
Needle-leaved-evergreen/deciduous forest	PFO4/1	0.5	-	<0.1	<0.1	0.3	0.6	1.8	4.0	7.9	19.2	65.6	35,521	14,381
Needle-leaved-/broad-leaved-evergreen forest	PFO4/3	0.2	-	-	-	2.5	1.3	1.3	11.1	18.9	12.2	52.5	410	166
Needle-leaved-evergreen/white-cedar forest	PFO4/8	-	-	-	-	-	0.0	0.8	2.6	18.4	28.6	49.7	368	149
Needle-leaved-evergreen forest/scrub-shrub	PFO4/PSS4	0.6	-	-	-	-	-	-	0.1	12.5	15.5	71.3	443	179
Dead trees	PFO5	-	-	-	-	-	-	-	-	28.1	-	71.9	10	4
White-cedar forest	PFO8	0.1	-	-	<0.1	0.1	0.8	0.7	2.6	5.5	14.8	75.3	12,162	4,924
White-cedar/deciduous forest	PFO8/1	0.3	-	-	<0.1	0.1	0.5	1.2	2.1	7.5	17.8	70.4	11,734	4,751
White-cedar/needle-leaved-evergreen forest	PFO8/4	0.1	-	-	-	-	0.7	2.9	3.2	6.7	21.2	65.2	596	241
White-cedar forest/dead trees	PFO8/5	-	-	-	-	-	-	-	-	-	8.3	91.7	1	0
White-cedar forest/scrub-shrub	PFO8/PSS8	-	-	-	-	-	1.7	1.2	0.5	12.5	26.1	58.1	187	76
Open water	POW	4.8	<0.1	0.8	1.9	3.8	5.1	5.5	8.2	12.6	19.8	37.5	5,619	2,275
Deciduous scrub-shrub	PSS1	2.8	<0.1	0.1	0.3	0.7	1.9	2.4	4.1	7.4	18.5	61.8	17,518	7,092
Deciduous/broad-leaved-evergreen scrub-shrub	PSS1/3	0.1	-	-	-	-	<0.1	0.3	2.5	5.0	18.1	74.0	826	335
Deciduous/needle-leaved-evergreen scrub-shrub	PSS1/4	0.7	-	-	<0.1	0.1	0.4	1.1	3.3	7.4	16.6	70.2	5,862	2,373
Deciduous/white-cedar scrub-shrub	PSS1/8	0.2	-	-	-	<0.1	-	1.2	2.9	7.4	22.0	66.3	1,890	765
Deciduous scrub-shrub/emergent (persistent vegetation)	PSS1/PEM1	0.1	-	-	0.8	2.0	2.7	1.6	3.1	8.3	16.1	65.3	1,357	549
Deciduous scrub-shrub/deciduous forest	PSS1/PFO1	0.1	-	-	0.1	<0.1	1.2	1.0	2.8	7.2	12.4	75.3	2,673	1,082
Broad-leaved-evergreen scrub-shrub	PSS3	33.2	-	-	-	-	-	0.1	0.6	1.8	5.9	58.4	898	363
Broad-leaved-evergreen/deciduous scrub-shrub	PSS3/1	0.0	-	-	-	-	-	-	0.2	4.7	4.0	91.1	420	170
Broad-leaved-/needle-leaved-evergreen scrub-shrub	PSS3/4	2.8	-	-	-	-	-	0.5	9.5	7.8	17.4	62.0	416	168
Needle-leaved-evergreen scrub-shrub	PSS4	0.5	-	<0.1	-	<0.1	0.2	1.0	1.8	5.9	11.1	79.5	3,617	1,464
Needle-leaved-evergreen/deciduous scrub-shrub	PSS4/1	0.8	-	-	-	<0.1	0.2	0.8	1.5	5.4	20.9	70.5	6,404	2,593
Needle-leaved-/broad-leaved-evergreen scrub-shrub	PSS4/3	-	-	-	-	-	-	-	1.1	9.0	-	89.9	175	71
Needle-leaved-evergreen/white-cedar scrub-shrub	PSS4/8	-	-	-	-	-	0.5	2.0	0.1	9.6	17.6	70.1	161	65
Needle-leaved-evergreen scrub-shrub/forest	PSS4/PFO4	0.3	-	-	-	-	0.6	1.4	4.4	5.6	2.0	85.7	852	345
White-cedar scrub-shrub	PSS8	0.1	-	-	-	0.1	0.2	0.6	3.7	8.2	26.5	60.5	791	320
White-cedar/deciduous scrub-shrub	`	0.2	-	-	-	-	0.3	0.1	1.3	3.3	16.2	78.6	2,856	1,156
White-cedar/needle-leaved-evergreen scrub-shrub	PSS8/4	-	-	-	-	-	-	<0.1	-	1.5	6.2	92.3	317	128
White-cedar scrub-shrub/forest	PSS8/PFO8	-	-	-	-	-	-	-	-	-	51.6	48.4	37	15
Lower-perennial stream	R2OW & R2UB2	0.1	<0.1	0.1	<0.1	1.2	2.6	3.8	6.0	12.1	25.6	48.5	581	235
Upper-perennial stream	R3OW & R3UB1	0.7	<0.1	0.2	0.5	2.5	5.0	5.3	4.3	13.8	32.3	35.4	321	130
Tidal water	Tidal Water	0.0	-	-	-	-	-	0.3	0.3	5.0	18.7	75.6	3,034	1,229
Richness (number of wetland types)		33	6	13	18	24	31	36	37	42	45	48	50	50

[1] Cowardin types classified as non-habitat under Anderson.

[2] All deciduous forest and deciduous scrub-shrub types are broad-leaved.

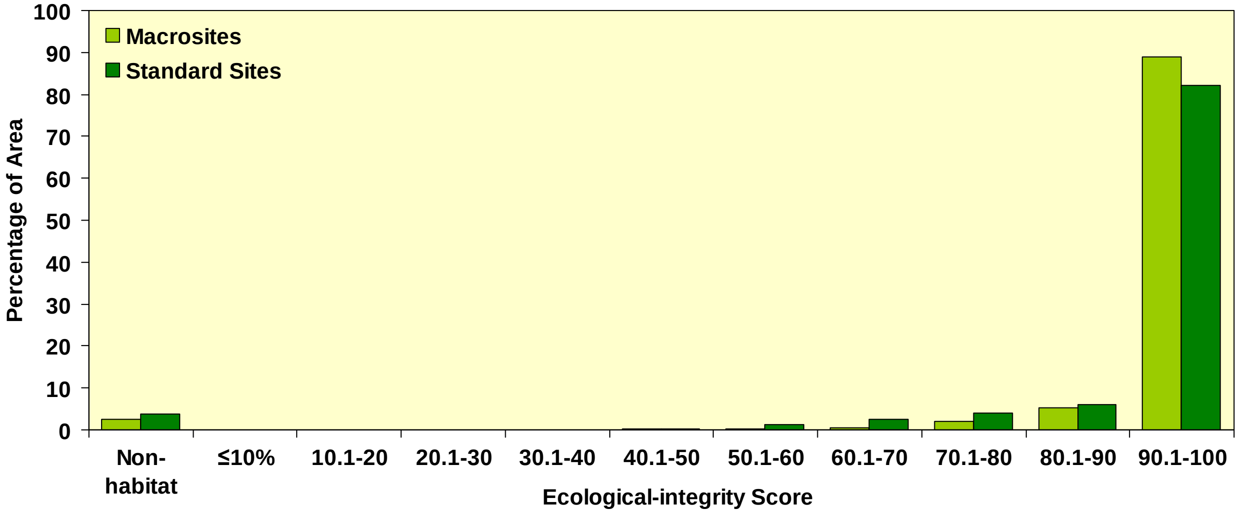

Figure 18. The percentage area of Office of Natural Lands Management priority sites associated with each of ten ecological-integrity classes.

Using point data provided by the New Jersey Endangered and Nongame Species Program, the distribution of threatened- and endangered-animal-species records was related to the ecological integrity of the associated Pinelands-habitat cells. For each record that fell on a non-habitat cell (e.g., dead-on-road pine snake), a landscape-integrity score was determined by measuring the extent of Pinelands habitat in the surrounding 1,000-m-radius neighborhood of the cell. This landscape-integrity score and the aquatic-integrity and wetland-drainage-integrity scores for the non-habitat cell were used to calculate an ecological-integrity score for that cell. Because the age of the data varied, each rare-species record was associated with ecological-integrity information from the most appropriate period. For example, pre-1986 species records were related to 1986 ecological-integrity data, 1990 records were related to an average ecological-integrity score based on 1986 and 1995 data, 2001 species records were related to an average score based on 1995 and 2002 data, and 2002 ecological-integrity scores were used for 2003 species records.

Pine snake (*Pituophis m. melanoleucus*) and Pine Barrens treefrog (*Hyla andersonii*) were the most frequently recorded species, accounting for a little more than one-half of all records. These two species, along with barred owl (*Strix varia*) and timber rattlesnake (*Crotalus horridus*), made up 79% of all data points. For most species, especially the more common ones, the number of records was proportional to the habitat area within each ecological-integrity class. With the exception of the red-headed woodpecker (*Melanerpes erythrocephalus*) and red-shouldered hawk (*Buteo lineatus*), the majority of records fell within the 80.1-90% and 90.1-100% ecological-integrity classes (Figure 19).

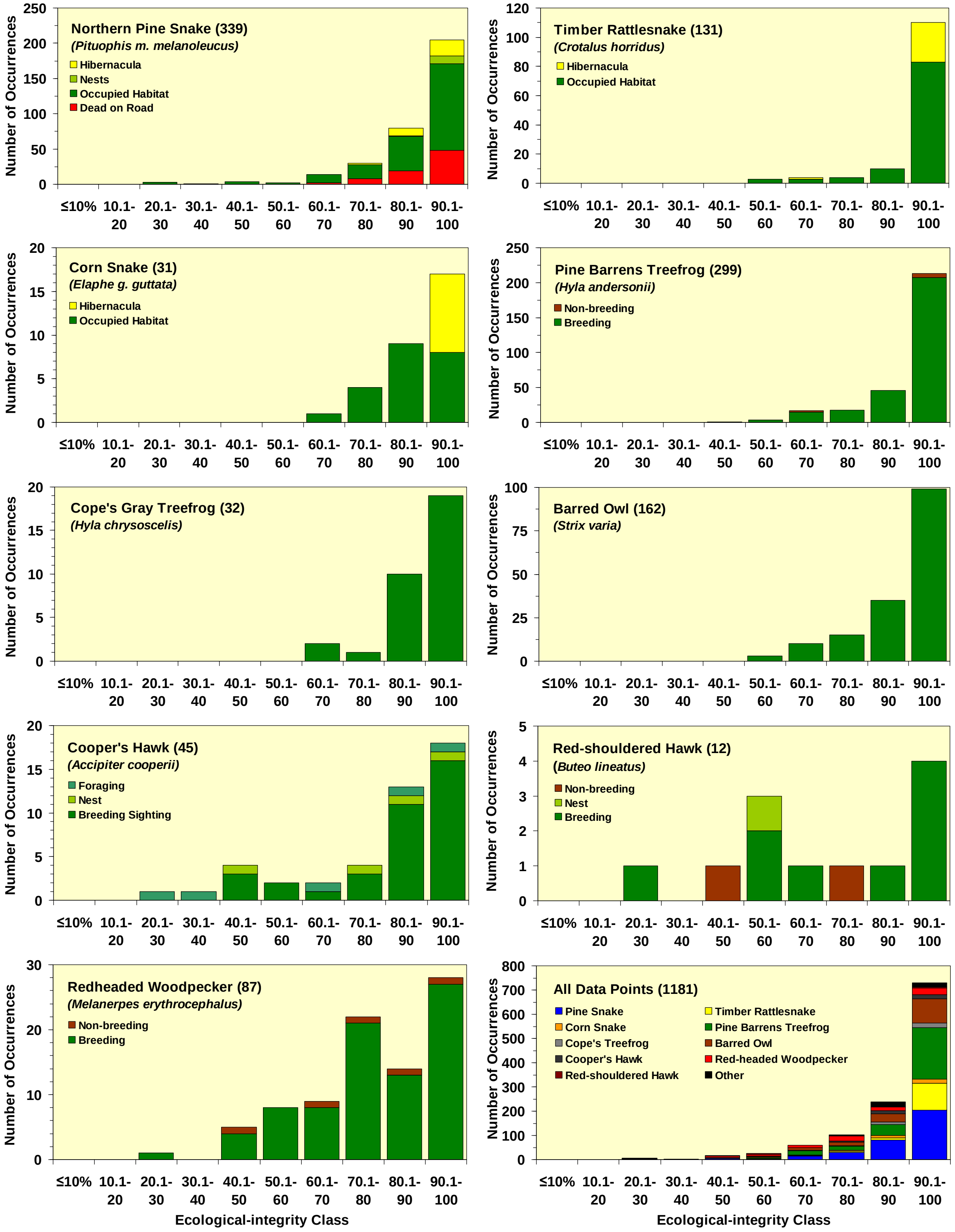

Figure 19. The number of threatened- or endangered-animal-species records in relation to the ecological-integrity of associated habitat or non-habitat cells.

CHANGES IN ECOLOGICAL INTEGRITY FROM 1986-1995 AND 1995-2002

The New Jersey Department of Environmental Protection land-use/land-cover data include two separate data sets. One integrates data from 1986 and 1995 (NJDEP 2000). The other integrates data from 1995 and 2002 (NJDEP 2007). Because the grain or spatial resolution of the integrated 1995-2002 data provides greater detail than the 1986-1995 data, separate change analyses were completed for the 1986-1995 and 1995-2002 periods. Changes occurring between 1986 and 2002 are given as estimates (Tables 7 and 8). The results of the transition analysis are presented with two caveats. First, some of the observed changes may be due to differences in interpretation between the different mapping periods. Secondly, a transition from one integrity class to another may involve small or large changes. For example, a change from the 80.1-90% integrity class to the 70.1-80% class may be due to a decrease of 1% (81 to 80%) or 10% (81 to 71%).

Table 7. Ecological-integrity transitions for 1986-1995 and 1995-2005. Values are acres of Pinelands habitat and non-habitat. Values above the shaded-diagonal cells represent an increase in integrity between periods. Values below the shaded-diagonal cells represent a decrease in integrity between periods. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

		1995 Ecological-integrity class											
		Non-habitat	≤10%	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1986 Total
1986 Ecological-integrity class	Non-habitat	156,482	-	87	237	292	649	671	932	1,264	1,914	1,513	164,041
	≤10	0	23	14	3	-	-	-	-	-	-	-	40
	10.1-20	30	5	698	165	-	-	-	-	-	-	-	898
	20.1-30	203	-	10	2,003	246	3	1	-	-	-	-	2,465
	30.1-40	307	-	-	241	4,706	464	<1	2	-	-	-	5,720
	40.1-50	509	-	-	<1	733	9,497	704	13	-	-	-	11,456
	50.1-60	1,079	-	-	<1	54	2,257	15,485	1,091	47	-	6	20,020
	60.1-70	2,093	-	-	-	130	394	4,577	28,449	1,734	106	43	37,526
	70.1-80	2,747	-	-	-	3	120	660	8,411	51,477	3,101	203	66,720
	80.1-90	3,697	-	-	<1	-	23	213	841	15,049	112,873	4,855	137,551
	90.1-100	2,413	-	-	-	-	<1	21	83	999	15,994	472,225	491,736
1995 Total		169,562	28	808	2,650	6,163	13,407	22,331	39,822	70,570	133,987	478,845	938,173

		2002 Ecological-integrity class											
		Non-habitat	≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1995 Total
1995 Ecological-integrity class	Non-habitat	162,215	4	97	134	254	391	460	544	573	997	1,337	167,006
	≤10	9	20	6	-	-	-	-	-	-	-	-	34
	10.1-20	118	34	601	41	3	-	-	-	-	-	-	798
	20.1-30	222	-	138	2,074	167	5	-	-	-	-	-	2,607
	30.1-40	431	12	8	367	4,905	219	1	-	-	-	-	5,943
	40.1-50	611	-	1	2	1,117	10,458	437	-	-	-	-	12,627
	50.1-60	1,095	-	1	5	59	2,446	17,656	550	9	3	-	21,824
	60.1-70	1,984	-	-	5	124	516	5,492	31,273	676	5	6	40,079
	70.1-80	2,354	-	-	-	17	84	538	9,347	56,376	1,512	34	70,261
	80.1-90	2,563	-	-	-	-	1	136	574	10,460	113,780	3,056	130,570
	90.1-100	1,330	-	-	-	-	2	9	51	476	12,069	472,488	486,424
2002 Total		172,931	70	853	2,629	6,646	14,122	24,729	42,339	68,569	128,364	476,921	938,173

Changes in ecological integrity occurring from 1986-1995 and 1995-2002 included both increases and decreases in integrity and conversion to non-habitat (Tables 7 and 8). Areas of non-habitat were also reclassified as habitat of varying integrity. The latter could be due to actual changes, differences in interpretation, or error, among other reasons. Although accounting for thousands of acres, changes in ecological integrity were relatively small compared to the extent of the Pinelands Area that remained unchanged (Tables 7 and 8). Except for the 90.1-100% ecological-integrity class, where relative changes were small, increases and decreases in ecological integrity and the amount of habitat converted to non-habitat during the 1986-1995 and 1995-2002 periods were proportional to the amount of habitat in the different integrity classes. Most losses of ecological integrity between periods were due to an area dropping to the next lowest integrity class. In both periods the total area that displayed a decrease in ecological integrity was much greater than the total area converted to non-habitat, indicating that loss of habitat has a pronounced affect on the ecological integrity of remaining habitat.

Table 8. Summary of ecological-integrity transitions for 1986-1995 and 1995-2002.

	Status of 1986 habitat and non-habitat in 1995					Acres in each integrity class		Net change (1986-1995)	
Ecological-integrity class	No change	Increased integrity	Converted to habitat	Decreased integrity	Converted to non-habitat	1986	1995	acres	%
Non-habitat	156,482	-	7,559	-	-	164,041	169,562	5,521	3.4
≤10%	23	17	-	-	0	40	28	-12	-30.5
10.1-20%	698	165	-	-5	-30	898	808	-90	-10.0
20.1-30%	2,003	249	-	-10	-203	2,465	2,650	185	7.5
30.1-40%	4,706	466	-	-241	-307	5,720	6,163	443	7.8
40.1-50%	9,497	717	-	-733	-509	11,456	13,407	1,951	17.0
50.1-60%	15,485	1,144	-	-2,311	-1,079	20,020	22,331	2,312	11.5
60.1-70%	28,449	1,883	-	-5,101	-2,093	37,526	39,822	2,296	6.1
70.1-80%	51,477	3,303	-	-9,192	-2,747	66,720	70,570	3,850	5.8
80.1-90%	112,873	4,855	-	-16,126	-3,697	137,551	133,987	-3,564	-2.6
90.1-100%	472,225	-	-	-17,098	-2,413	491,736	478,845	-12,892	-2.6
1995 Total	853,916	12,800	7,559	-50,818	-13,080	938,173	938,173	0	0

	Status of 1995 habitat and non-habitat in 2002					Acres in each integrity class		Net change (1995-2002)	
Ecological-integrity class	No change	Increased integrity	Converted to habitat	Decreased integrity	Converted to non-habitat	1995	2002	acres	%
Non-habitat	162,215	-	4,792	-	-	167,006	172,931	5,925	3.5
≤10%	20	6	-	-	-9	34	70	35	102.8
10.1-20%	601	44	-	-34	-118	798	853	55	6.9
20.1-30%	2,074	172	-	-138	-222	2,607	2,629	22	0.8
30.1-40%	4,905	220	-	-387	-431	5,943	6,646	703	11.8
40.1-50%	10,458	437	-	-1,120	-611	12,627	14,122	1,495	11.8
50.1-60%	17,656	562	-	-2,511	-1,095	21,824	24,729	2,905	13.3
60.1-70%	31,273	686	-	-6,137	-1,984	40,079	42,339	2,260	5.6
70.1-80%	56,376	1,546	-	-9,986	-2,354	70,261	68,569	-1,692	-2.4
80.1-90%	113,780	3,056	-	-11,171	-2,563	130,570	128,364	-2,206	-1.7
90.1-100%	472,488	-	-	-12,606	-1,330	486,424	476,921	-9,503	-2.0
2002 Total	871,845	6,729	4,792	-44,090	-10,717	938,173	938,173	0	0

Changes in ecological integrity between periods were summarized by management area. The largest net increase in non-habitat during both periods occurred in Regional Growth Areas and Rural Development Areas (Figure 20). The greatest net decrease in the ecological integrity of existing habitat in the top three ecological-integrity classes also occurred in these two management areas. The greatest net decrease in the 90.1-100% integrity class occurred in Forest Areas. The large increase in the 80.1-90% class in the Forest Area occurred primarily at the expense of the 90.1-100% class. Decreases in ecological integrity and conversion of habitat to non-habitat during both periods generally occurred adjacent to existing non-habitat (Figures 21 and 22).

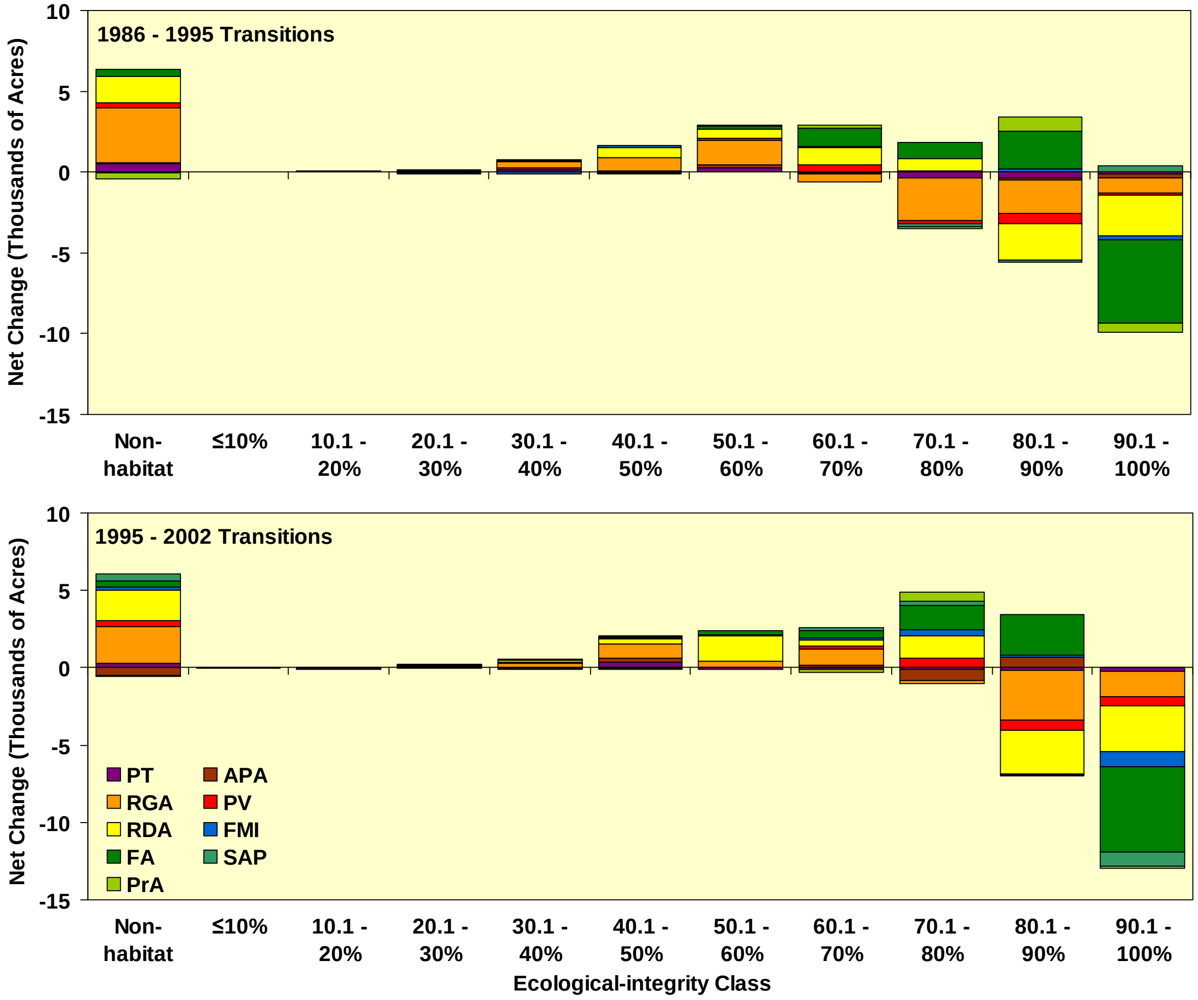

Figure 20. Net changes in the acres of habitat in each ecological-integrity class by Pinelands management area for the periods 1986-1995 and 1995-2002. Stacked bars show the nature of the change. For example, the largest increase in non-habitat during both periods occurred in Regional Growth Areas and Rural Development Areas.

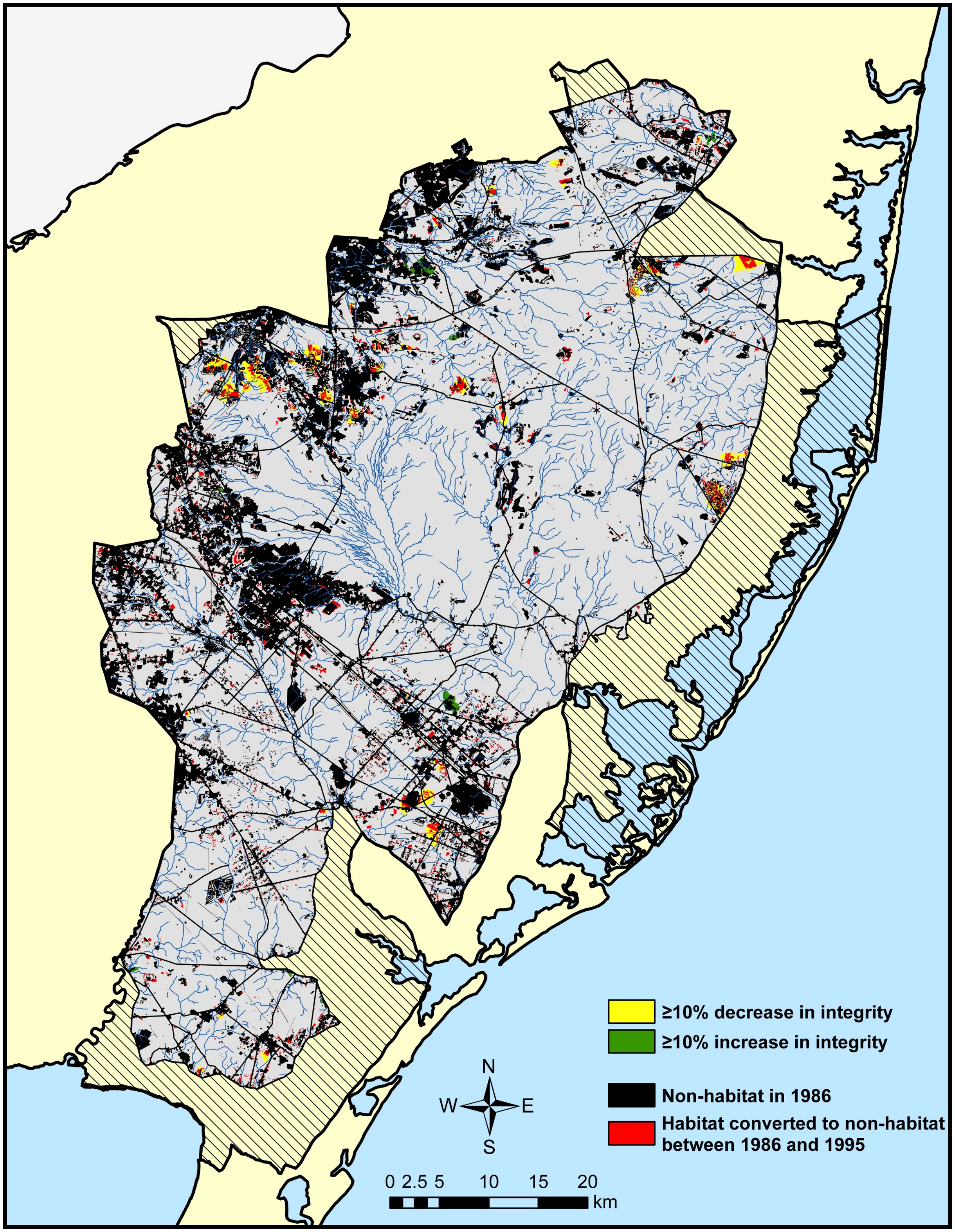

Figure 21. Areas of Pinelands habitat that displayed at least a ten percent increase or decrease in ecological integrity between 1986 and 1995. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

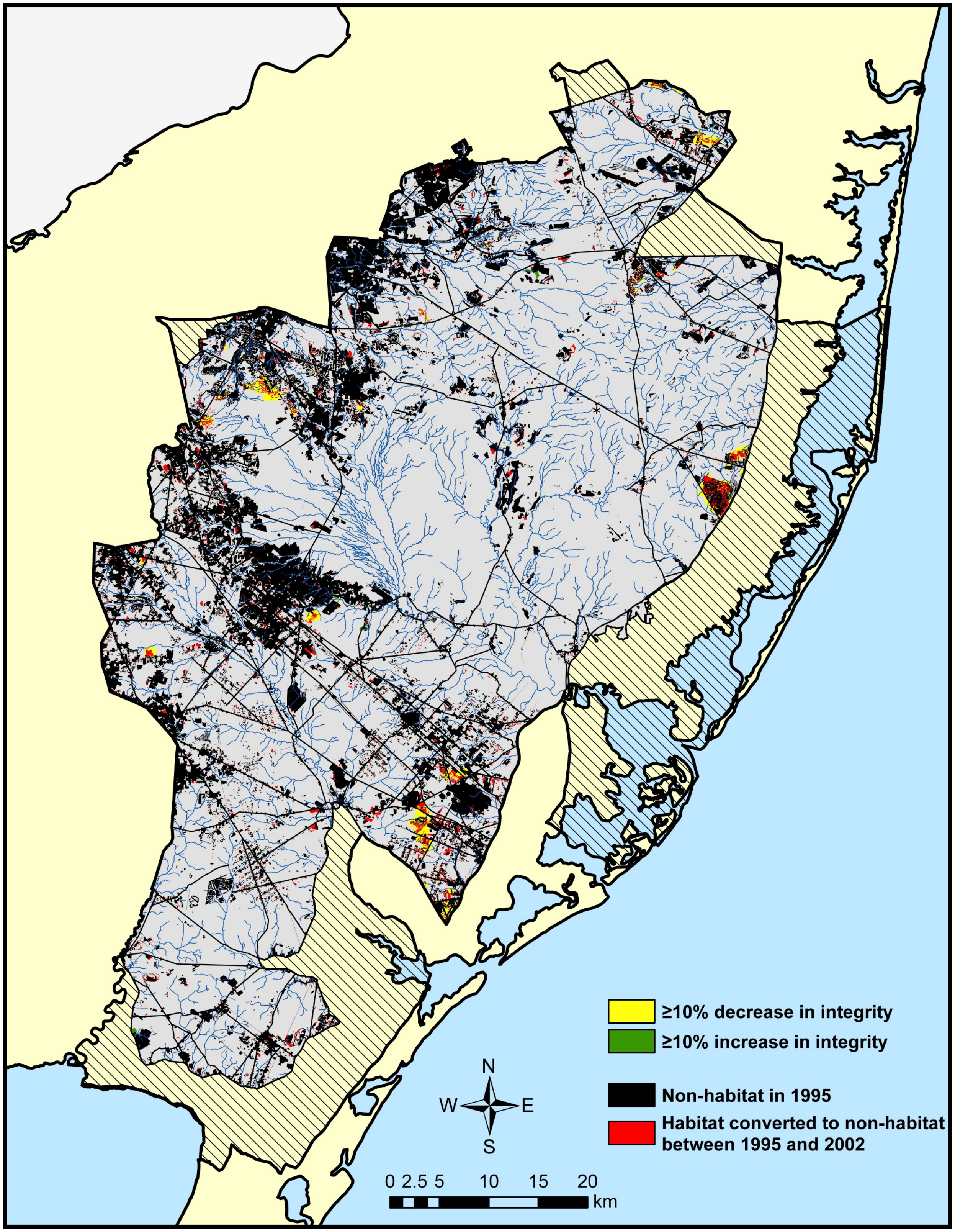

Figure 22. Areas of Pinelands habitat that displayed at least a ten percent increase or decrease in ecological integrity between 1995 and 2002. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

SIZE-CLASS DISTRIBUTION OF DISCRETE AREAS OF ECOLOGICAL INTEGRITY

Adjacent and diagonal 10×10-m Pinelands-habitat cells within the same ecological-integrity class were clumped using Patch Analyst for Grids (Centre for Northern Forest Ecosystem Research 2008). The clumps represented groups of habitat cells with similar integrity values rather than physical features of the landscape. A total of 12,159 clumps, ranging in size from 0.0247 to 314,400 acres (0.01 to 127,287 ha), were produced and categorized using nine clump-size classes (Tables 9 and 10).

The contribution of clumps <25 acres to the area within individual ecological-integrity classes decreased with increasing integrity (Figure 23). Conversely, the contribution of clumps ≥250 acres increased along the integrity gradient. Although 98% of all clumps were <250 acres, 81% of Pinelands habitat consisted of clumps ≥250 acres, with 68% of the habitat comprised of clumps ≥1,000 acres. Nearly all of the area in the ≥1,000-acre clump-size class fell within the 80.1-90% and 90.1-100% ecological-integrity classes. About forty-four percent of all clumps were <1 acre, but these small clumps accounted for only 0.2% of the Pinelands Area. Single 10×10-m Pinelands-habitat cells represented 21% of all clumps that were less than one acre.

The apparent isolation of small clumps was partly attributed to the use of integrity classes and raster data. The classification created abrupt boundaries where a gradual change in integrity existed. A difference of a tenth of a percent or less in ecological-integrity scores can place two adjacent cells in different integrity classes. A small clump can also be separated from a larger clump of similar integrity by only a few cells or even a single cell of different integrity.

Application of the ecological-integrity-assessment in land-use planning might require that boundaries or gaps between clumps of varying integrity be evaluated further. Will the merger of small clumps with different integrity values facilitate planning? Is the transition between clumps abrupt or gradual? How far apart are separate clumps of equal integrity? What are the regional-conservation implications of merging a high-quality clump with one of lower quality with the intent of either allowing more intensive land use on the clump or reducing the intensity of permitted uses? What amount of high- or low-integrity habitat might be affected by a merger of habitat cells? These are only a few of the questions that can be addressed on a case-by-case basis with the ability to assess the overall, cumulative effect on the status of the Pinelands Area. In effect, each question deals with policy rather than science, although the consequences of all policy decisions concerning ecological integrity can be quantified using the assessment-data layers and a geographic information system.

Tables 11 and 12 provide two quantitative examples of the result of merging clumps with different integrity-classes. In one example, clumps with integrity scores that ranged from 70.1-100% were merged. In the second example, clumps with integrity scores that ranged from 80.1-100% were merged. With the exception of the largest clumps whose total acreage increased, the area within each clump-size class decreased dramatically, with an average decrease of 74% for the 80.1-100% merger and an average decrease of 84% for the 70.1-100% merger. The number of clumps associated with each integrity class also decreased considerably following the merger, with the number of clumps decreasing by an average of 73% for the 80.1-100% merger and 83% for the 70.1-100% merger.

Table 9. Acres of integrity-clump-size classes in each ecological-integrity class.

Ecological-integrity class	Clump size (acres)									
	<1	1-9	10-24	25-49	50-99	100-249	250-499	500-999	≥1,000	Total acres
≤10%	9	43	18	-	-	-	-	-	-	70
10.1-20%	42	435	313	63	-	-	-	-	-	853
20.1-30%	93	1,000	611	369	309	246	-	-	-	2,629
30.1-40%	170	1,495	1,610	1,672	1,365	334	-	-	-	6,646
40.1-50%	234	2,080	2,577	2,755	3,284	2,530	663	-	-	14,122
50.1-60%	257	2,603	3,192	4,752	5,161	5,860	2,254	649	-	24,729
60.1-70%	269	2,486	3,481	4,982	8,275	11,050	5,357	5,166	1,274	42,339
70.1-80%	231	1,998	3,138	4,179	8,663	18,149	14,658	9,745	7,808	68,569
80.1-90%	169	1,289	1,981	3,852	5,165	16,537	20,341	28,882	50,149	128,364
90.1-100%	46	406	594	726	1,014	3,120	2,899	5,120	462,996	476,921
Total acres	1,521	13,835	17,515	23,351	33,236	57,824	46,172	49,561	522,227	765,242
Percent of total	0.2	1.8	2.3	3.1	4.3	7.6	6	6.5	68.2	100

Table 10. The number of integrity clumps in each clump-size class by ecological-integrity class.

Ecological-integrity class	Clump size (acres)									
	<1	1-9	10-24	25-49	50-99	100-249	250-499	500-999	≥1,000	Total clumps
≤10%	28	15	1	-	-	-	-	-	-	44
10.1-20%	126	132	23	2	-	-	-	-	-	283
20.1-30%	281	295	43	11	5	2	-	-	-	637
30.1-40%	499	436	99	48	20	3	-	-	-	1,105
40.1-50%	753	589	160	77	47	18	2	-	-	1,646
50.1-60%	914	734	197	135	72	38	6	1	-	2,097
60.1-70%	988	681	216	135	116	74	17	8	1	2,236
70.1-80%	948	554	193	118	123	119	43	15	6	2,119
80.1-90%	678	359	121	104	73	104	57	42	22	1,560
90.1-100%	193	118	35	20	15	20	8	7	16	432
Total number	5,408	3,913	1,088	650	471	378	133	73	45	12,159
Percent of total	44.5	32.2	8.9	5.3	3.9	3.1	1.1	0.6	0.4	100

Table 11. Acres of integrity-clump-size classes in merged ecological-integrity classes and the percentage change in the clump area of each clump-size class following the merger.

Ecological-integrity class	Clump size (acres)									
	<1	1-9	10-24	25-49	50-99	100-249	250-499	500-999	≥1,000	Total acres
70.1–100%	171	1,123	1,082	851	1,203	2,399	1,863	6,481	658,681	673,855
% Change	-62%	-70%	-81%	-90%	-92%	-94%	-95%	-85%	26%	0%
80.1–100%	105	706	864	950	1,100	1,910	3,896	6,193	589,562	605,285
% Change	-51%	-58%	-66%	-79%	-82%	-90%	-83%	-82%	15%	0%

Table 12. The number of integrity clumps in each clump-size class in merged ecological-integrity classes and the percentage change in the number of clumps in each clump-size class following the merger.

Ecological-integrity class	Clump size (acres)									
	<1	1-9	10-24	25-49	50-99	100-249	250-499	500-999	≥1,000	Total clumps
70.1-100%	623	380	72	24	18	16	6	9	16	1,164
% Change	-66%	-63%	-79%	-90%	-91%	-93%	-94%	-86%	-64%	-72%
80.1-100%	402	226	54	26	16	12	11	9	15	771
% Change	-54%	-53%	-65%	-79%	-82%	-90%	-83%	-82%	-61%	-61%

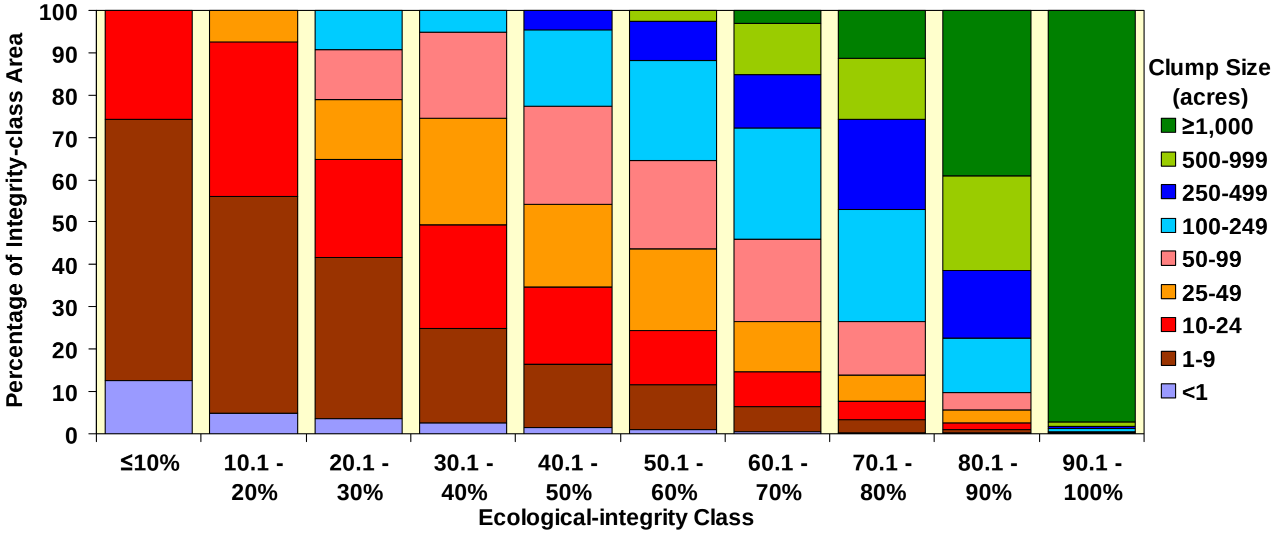

Figure 23. The percentage of each ecological-integrity class represented by clumps ranging in size from <1 acre to ≥1,000 acres.

APPENDIX 1. A REVIEW OF THE LANDSCAPE, AQUATIC, AND WETLAND ECOLOGY LITERATURE

LANDSCAPE ECOLOGY

The Pinelands Landscape

Extensive upland and wetland habitats characterize the natural landscape of the Pinelands (Forman 1979a). Following McCormick's (1979) concise and widely used classification of the many possible upland- and wetland-habitat types, uplands include pine-oak forests, which are dominated by pitch pine (*Pinus rigida*) and classified according to the abundance of associated oak species such as blackjack oak (*Quercus marilandica*), black oak (*Quercus velutina*) and white oak (*Quercus alba*), and oak-pine forests dominated by one or more tree-form oak species. McCormick (1979) included the Pine Plains as an extreme example of the pine-blackjack oak forest type (McCormick and Buell 1968, Good et al. 1979, Gibson et al. 1998). McCormick’s (1979) wetland types are southern (Atlantic) white cedar (*Chamaecyparis thyoides*) swamp forests (Little 1950, 1951, Ehrenfeld and Schneider 1991, Laidig and Zampella 1999), broadleaf or hardwood swamp forests (Bernard 1963, Olsson 1979, Ehrenfeld and Gulick 1981, Ehrenfeld 1986), pitch pine lowland and pine transition forests (Zampella et al. 1992), shrubby wetland communities (Olsson 1979), and herbaceous wetland communities, including savannas (Harshberger 1916) and vegetation types associated with streams (Morgan and Philipp 1986, Zampella and Laidig 1997) and ponds (Zampella and Laidig 2003).

In 2002, upland and wetland/water habitats covered 54% and 27% percent of the Pinelands Area, respectively (Figure 1.1). Most Pinelands wetlands are associated with streams networks, although some, such as pitch pine lowland complexes, occur as broad, poorly drained plains that extend far beyond stream networks. Wetlands represent relatively unbroken habitats that provide spatial continuity of the natural Pinelands landscape throughout the region.

Ecological Mosaics

The Pinelands landscape represents an ecological mosaic composed of different habitat patches that reflect variations in topography, soils, hydrology, and natural and human-related disturbance history, among other factors (Figure 1.2, Forman 1979b). Forman (1995) defined patch as “a relatively homogeneous nonlinear area that differs from its surroundings,” adding that the “internal micro heterogeneity present is repeated in similar form throughout the area of a patch.” In practice, the term patch is sometimes applied to more heterogeneous habitats surrounded by human-dominated landscapes.

Patches are embedded in a background matrix that is characterized by extensive cover and high connectivity (Forman 1995). A matrix may be composed of many cover types (Ricketts 2001). Connectivity, which is a measure of spatial continuity (Forman 1995), can be viewed as functional or structural. Functional connectivity is species-specific and deals with the movement of organisms of interest across the landscape, whereas structural connectivity is simply a measure of habitat contiguity (Taylor et al. 1993, Tischendorf and Fahrig 2000). Functional connectivity depends on landscape composition, landscape configuration, and the ease of movement of individuals through the matrix (Taylor et al. 1993, Murphy and Lovett-Doust 2004), and can be influenced by habitat quality (Bowne et al. 2006), matrix characteristics (Ricketts 2001, Castellón and Sieving 2006), and loss of habitat, regardless of the spatial configuration of the loss (D’Eon et al. 2002).

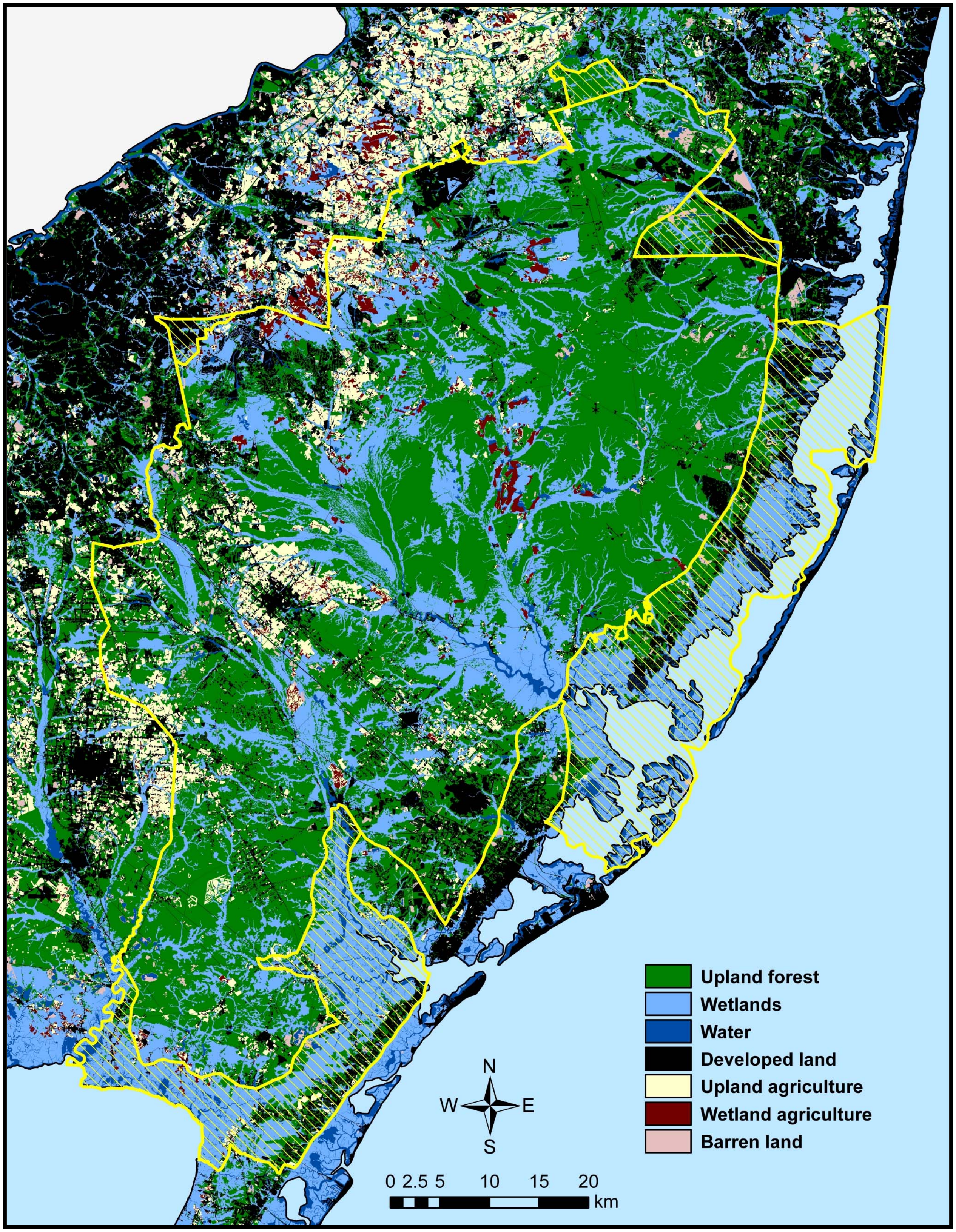

Figure 1.1. Pinelands Area land-use/land-cover in 2002 (NJDEP 2007). The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

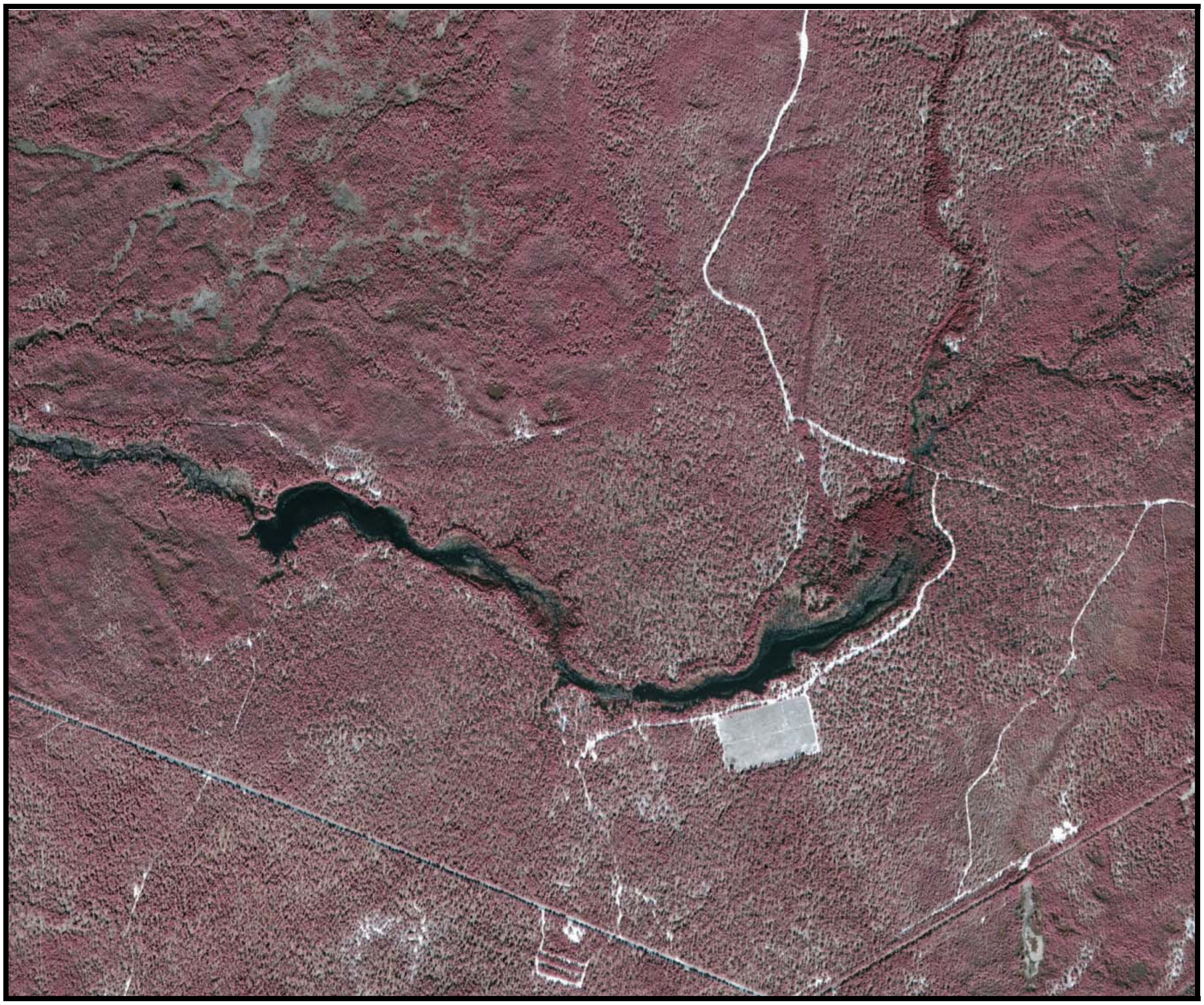

Figure 1.2. A mosaic of upland and wetland habitats in the Skit Branch watershed, Wharton State Forest, Tabernacle, NJ.

Forman (1995) defined corridor as a strip of land that differs from adjacent lands and considered connectivity to be an inherent structural attribute of corridors whether or not the corridor promotes movement among patches. Fahrig (2003) suggested that the amount of habitat affects movement between patches and that the effects of landscape-scale habitat amount and patch isolation, which is the degree of separation between patches, are the same.

Throughout most of the Pinelands, extensive upland-forest and wetland habitats represents the matrix in which patches of developed and agricultural land are embedded. Structural connectivity is high in these areas. In heavily altered areas of the Pinelands, forest remnants (Forman 1979b) represent patches within an urban and agricultural matrix. In these heavily altered landscapes, the matrix represents non-habitat surrounding natural-habitat patches (Ricketts 2001, Murphy and Lovett-Doust 2004, Bender and Fahrig 2005). Structural connectivity in the heavily altered areas of the Pinelands can be low, although wetlands and narrow upland-forest corridors may connect natural-habitat fragments.

The Ecological Significance of Patch Size and Habitat Area

Studies of terrestrial birds in habitat islands were among the earliest investigations that contributed to our understanding of the relationship between habitat area and regional biota. Many studies found that the total number of bird species in forest patches increased with area (Galli et al. 1976, Forman et al. 1976, Martin 1981, Ambuel and Temple 1983, Blake 1983, Howe 1984, Freemark and Merriam 1986, Freemark and Collins 1989, Blake and Karr 1987).

Although bird-species richness generally increases with forest-island size, this relationship is complicated by other factors such as habitat heterogeneity and regional-landscape structure (Lynch and Whigham 1984, Freemark and Merriam 1986, Boecklen 1986, Blake and Karr 1987, Lee et al. 2002). In many regions, habitat area might explain only about 50% of the variation in species richness (Boeklen and Gotelli 1984, Blake and Karr 1987). A large forest generally supports more bird species than a comparable area of small forests, but there are cases where this relationship may not hold (Blake and Karr 1987).

Wander (1980) found that habitat area was a less important determinant of bird-species diversity in Pinelands cedar swamps than vegetation composition. Lynch and Whigham (1984) estimated the abundance and diversity of forest birds in 270 upland forest patches in Maryland’s Coastal Plain Province in relation to patch size, degree of isolation, and habitat characteristics. Habitat characteristics rather than patch size appeared to play the dominant role in determining community composition and local abundance of bird species. Lynch and Whigham (1984) attributed the weak relationship between species richness and patch size to the relatively low degree of isolation of forest patches on the Maryland Coastal Plain, a condition similar to that found in parts of the Pinelands. Blake and Karr (1987) suggested that species-area relationships might have greater utility in regions where habitat patches are well isolated and contrast sharply with the surrounding matrix. Such conditions may exist in heavily altered areas of the Pinelands.

Species-area studies have revealed that Neotropical migrants are especially sensitive to forest-patch size and that their decline in North America could be attributed to habitat loss and fragmentation (Whitcomb et al. 1981, Ambuel and Temple 1983, Howe 1984, Blake and Karr 1984, Robbins et al. 1989a, Askins et al. 1990, Donovan et al. 1995, among many others), in addition to several other factors such as changes beyond the breeding range (Whitcomb et al. 1981, Ambuel and Temple 1982, Hall 1984, Askins et al. 1990), cowbird (*Molothrus ater*) nest parasitism (Whitcomb 1977, Brittingham and Temple 1983, Donovan et al. 1995, Robinson et al. 1995) and nest predation (Whitcomb 1977, Wilcove 1985, Böhning-Gaese et al. 1993, Bollinger and Linder 1994, Donovan et al. 1995) within the breeding range.

Blake and Karr (1984) noted that although two small forest reserves in their Illinois study area supported more bird species than one large reserve of equal size, a single large reserve was more likely to support a greater number of long-distance migrants and forest-interior species. Flather and Sauer (1996) reported that in the eastern United States, Neotropical migrants were generally more abundant in landscapes with larger forest patches, a greater proportion of forest and wetland habitats, and fewer edge habitats, but cautioned about extrapolating results from one landscape to others. In their study of a Canadian agricultural landscape, Lee et al. (2002) found that the relationship between bird-species abundances, patch size, within-patch habitat, and surrounding forest cover varied among three Neotropical species. Ovenbird (*Seiurus aurocapillus*) abundance was related to both patch size and surrounding forest cover, although surrounding forest cover explained more of the variation in ovenbird abundance than did patch size. Red-eyed vireo (*Vireo olivaceus*) and wood thrush (*Hylocichla mustelina*) abundances were related to deciduous-forest cover within patches and patch size rather than to surrounding forest cover. In another Canadian study, Burke and Nol (2000) found that adult-female reproductive success of ovenbirds, wood thrushes, rose-breasted grosbeaks (*Pheucticus ludovicianus*), and veeries (*Catharus fuscescens*) was at or above replacement levels in both continuous forest and woodlots averaging 849 ha (2,097 acres) and below replacement levels in woodlots averaging 93 ha (230 acres). Red-eyed vireos were close to replacement levels only in continuous forest. Local forest cover had little additional effect on the reproductive success of these species. Hoover et al. (1995) also found that wood thrush nesting success was highest in contiguous forest and lowest in forest patches <80 ha (198 acres).

Darr et al. (1998) reported that the probability of occurrence of 24 area-sensitive breeding-bird species in Maryland's Coastal Plain increased with forest area. For most of these species the probability of occurrence also increased with the total area of forest within 1 km (0.62 mi) of a survey point. This information was used by Darr et al. (1998) to identify priority conservation areas and sites where development could occur. The association between the occurrence of area-sensitive birds and surrounding forest was also studied by Boulinier et al. (2001), who found that species richness was lower and mean year-to-year extinction rates were higher on North American Breeding Bird Survey routes surrounded by landscapes with a lower mean-patch size. Reduced forest cover can also influence the movement of some bird species. In a field experiment in a Canadian agricultural landscape, three species of territorial, mated-male forest-bird species were moved from their territories and released (Bélisle et al. 2001). All three species took more time and were less likely to return to their territories as forest cover decreased in the landscape.

Keller et al. (1993) studied the relationship between riparian-forest width and bird-species composition in agricultural landscapes of the Delmarva Peninsula. Several area-sensitive Neotropical migrant species were encountered more frequently in wider riparian forests. Although Keller et al. (1993) recommended that riparian corridors at least 100 m (328 ft) wide be provided to function as habitat for forest-interior birds, they recognized that wider corridors would be preferable since they are most likely to provide habitat for forest-interior birds and have less forest edge. Hodges and Krementz (1996) found that species richness was lower in narrow (<350 m or 1,148 ft) riparian forests compared to medium (400-700 m or 1,312-2,297 ft) and wide (>1000 m or 3,281 ft) riparian forests bordered by pine plantations along Georgia's Altamaha River and that the probability of encountering five of the six most-abundant Neotropical species increased with forest corridor width. In South Carolina, Kilgo et al. (1998) reported that total bird-species richness and Neotropical-species richness increased as the width of bottomland hardwood stands increased, even though the adjacent habitat was also forested. They suggested that although narrow-riparian stands provide valuable bird habitat, maintaining the full complement of bird species characteristic of the region's bottomland hardwoods requires conservation of wide riparian zones.

Findlay and Houlahan (1997) studied the relationship between bird- and mammal-species richness, wetland area, and surrounding landscape in Canadian wetlands. Bird-species richness increased with wetland area and decreased with increasing paved-road density on adjacent lands up to 2 km (1.2 mi) from the wetlands. Mammal-species richness increased with wetland area and surrounding forest area. Studies conducted in Indiana produced different results concerning the effect of habitat area on small mammals. In one study (Gottfried 1979), forest habitats over 30 ha (74 acres) in size supported more small-mammal species at higher densities compared to woodlots surrounded by cornfields. White-footed mouse (*Peromyscus leucopus*) densities, which were higher in forest habitats, were inversely related to isolation. In contrast, Nupp and Swihart (1996, 1998) reported that white-footed-mouse densities were inversely related to forest area. Bayne and Hobson (1998) found that patch size and edge had little effect on small-mammal abundances, with the exception of deer mice (*Peromyscus maniculatus*), which were more abundant at the edge of Canadian farm woodlots compared to woodlot interiors.

Patch size can also influence plant-community composition and affect several taxonomic groups within the same habitat. Houlahan et al. (2006) found that total plant-species richness in

Canadian wetlands was positively correlated with wetland area and the amount of upland forest on adjacent lands. Species richness of birds, mammals, herptiles (reptiles and amphibians), and plants in Canadian wetlands also increased with wetland area, herptile- and mammal-species richness increased with an increase in surrounding forest area, and plant-, bird-, and herptile-species richness decreased with an increase in paved-road density on adjacent lands up to 2 km away (Findlay and Houlahan 1997). Bird-, amphibian-, reptile-, and mammal-taxonomic richness in fragmented sand-pine-scrub habitat in Florida was correlated with habitat area, although combinations of small (≤ 10 ha or 24.7 acres) and medium (25-50 ha or 62-124 acres) scrub habitats supported more taxa than individual large (170-190 ha or 420-469 acres) scrub habitats (McCoy and Mushinsky 1994).

Habitat Loss and Fragmentation

Fragmentation is the breaking up of a habitat or land type into smaller parcels, with smaller parcels becoming somewhat widely and usually unevenly separated (Forman 1995). Although this definition of fragmentation includes any kind of land cover, it is generally used to describe the breaking up of natural habitat. Fragmentation can be a result of natural or quasi-natural disturbances such as wind, wildfire, flooding, and outbreaks of herbivores or pathogens or can be solely the result of human activities (Dale et al. 2000). The process increases the number of patches and total edge length and decreases the average patch size, the total amount of interior or core habitat, and connectivity across an area (Forman 1995). Fragmentation effects are highly specific to the species, scale, and processes studied (Debinski and Holt 2000, Villard 2002).

Although habitat loss and fragmentation are distinctly different phenomena (Fahrig 1997, McGarigal and Cushman 2002), in much of the scientific literature the term fragmentation has been applied to both habitat loss and the breaking up of habitat (Wilcove et al. 1986, Andrén 1994, Forman 1995, Fahrig 2003). This dual meaning is confusing since both processes can occur independently and have very different ecological effects (Forman 1995, Fahrig 2003). Since fragmentation usually occurs through a process of habitat loss and the breaking apart of habitat, the effects of the two processes are confounded in studies that do not account for habitat loss separately (Flather and Bevers 2002, Fahrig 2003). Fahrig (2003) described some typical relationships between the proportion of habitat and common measures of fragmentation, including linear (mean patch size), bell-shaped (number of patches and total edge), sigmoidal (size of largest patch), or negative-exponential (mean nearest-neighbor distance) relationships, which highlights the ambiguity between habitat loss and fragmentation.

In many habitat-fragmentation studies, the effects of habitat loss are attributed to fragmentation (Fahrig 2003). In a review of the effects of habitat fragmentation on birds and mammals, Andrén (1994) found that in landscapes with more than 30% of suitable habitat, fragmentation is primarily habitat loss, whereas in landscapes with highly fragmented habitats, patch size and isolation complement the effect of habitat loss, resulting in a loss of species or a decline in population size greater than what would be expected from habitat loss alone. He also noted that the configuration of habitat is less important in landscapes with a high proportion of suitable habitat, with the opposite being true in highly fragmented landscapes.

Boulinier et al. (1998), who characterized fragmentation as the average forest-patch size in circular areas centered on breeding-bird-survey routes, found that forest-bird-species richness increased with increasing forest-patch size. Based on a review of 25 studies examining the effect of patch size on population densities of insects, mammals, and birds, Bender et al. (1998) reported that increasing patch size had a negative effect on edge species, a positive effect on interior species, and a negligible effect on generalist species that used both edge and interior habitats. Patch-size effect was lower for migratory species compared to resident species. They predicted that in landscapes undergoing habitat loss and fragmentation, the decline in interior-species-population densities associated with fragmentation would be greater than that predicted from habitat loss alone and that the opposite effect would be observed for edge species. Trzcinski et al. (1999) concluded that the effect of forest cover on the distribution of forest-interior nesting birds is greater than that of forest fragmentation. They suggested that the extent of habitat is more important than spatial patterns and that the primary focus of conservation should be on preventing a decrease in forest cover. Likewise, the available empirical evidence suggests that habitat loss has a large negative effect on biodiversity and that fragmentation independent of habitat loss has weaker effects that may be positive or negative (Fahrig 2002, 2003). Fahrig (1997) suggested that habitat spatial patterns are unlikely to mitigate the risks of habitat loss, an opinion supported by Harrison and Bruna (1999) in their assessment of the role of corridors, and that conservation efforts aimed at endangered species should be directed towards stopping habitat loss and increasing habitat restoration efforts.

Kolozsvary and Swihart (1999) surveyed amphibians in an agricultural landscape in Indiana where riparian areas or small woodlot fragments were surrounded by farmlands. Forest covered only 16% of the study area. They concluded that different amphibian species responded differently to fragmentation in a manner that reflects differences in life history. Forest dependent species such as wood frogs (*Rana sylvatica*) and redback salamanders (*Plethodon cinereus*) were most sensitive to a reduction in forest area, whereas American toads (*Bufo americanus*) and northern gray treefrogs (*Hyla versicolor*), which were widely distributed, appeared to be successful in agricultural landscapes. The occurrence of tiger salamanders (*Ambystoma tigrinum*) and green frogs (*Rana clamitans melanota*) was related to the proximity of suitable breeding habitat.

For mammals, field studies have shown that diversity varies across habitats and landscapes and that the effect of forest-habitat loss and fragmentation on mammals is variable and not always negative. Nupp and Swihart (2000) reported that although the response of small mammals to agriculturally induced forest fragmentation varied among species, the number of forest-dwelling, small-mammal species increased with forest area and was highest in continuous-forest sites. In a study of experimentally created habitat fragmentation represented by a series of mowed and unmowed patches, Bowers and Dooley (1999) found that fragmentation had a beneficial effect on meadow voles (*Microtus pennsylvanicus*). In another experimental study, white-footed mouse densities were highest on small old-field patches (Foster and Gaines 1991).

In the Pinelands, fragmentation and loss of habitat is associated with natural factors such as flooding due to beaver activity (Zampella and Lathrop 1997), quasi-natural wildfires (Little 1979, Buchholz and Zampella 1987, Forman and Boerner 1981), historic resource exploitation (Wacker 1979), and urban and agricultural land uses (Bunnell et al. 2003, Luque et al. 1994). Luque et al. (1994) described landscape changes in the northeastern part of the Pinelands between 1972 and 1988. During this period, the number of forest patches increased and forest-patch size decreased. The opposite trend was observed for non-forest habitats, which included urban-land uses and transitory habitats associated with wildfire and timber harvesting. Bunnell et al. (1999) quantified changes in patch size and number in the Mullica River Basin between 1979 and 1991 and reported similar trends. The total area and number of developed-land, managed-grassland, and barren-land patches increased during the 12-year study period. An increase in the number of forest patches and a decrease in the total area and median size of forest patches indicated that loss and fragmentation of forest occurred during the study period. Loss of forest to development and associated cover types was a major land-use transition.

Between 1986 and 2002, developed land increased from about 8.5% to 10.4% of the 938,173-acre (379,827-ha) Pinelands Area. About 2% of Pinelands habitat present in 1986 was converted to developed land during this period. The majority of the 17,142-acre (6,940-ha) net increase in developed land was associated with the loss of upland forest (Table 1.1). Most of the habitat loss and associated fragmentation occurred along the periphery of the Pinelands Area (Figure 1.3). Sixty-two percent of the habitat converted to developed land occurred within 100 m (328 ft) of existing development, with 95% of the conversion found within 400 m (1,312 ft) of existing developed land.

Table 1.1. Development related land-use transitions from 1986-1995 and 1995-2002. Two separate transition periods are given due to differences in mapping resolution between the periods. The New Jersey Department of Environmental Protection land-use/land-cover data include two separate data sets. One integrates data from 1986 and 1995 (NJDEP 2000). The other integrates data from 1995 and 2002 (NJDEP 2007). Because the grain or spatial resolution of the integrated 1995-2002 data provides greater detail than the 1986-1995 data, separate change analyses were completed for the 1986-1995 and 1995-2002 periods.

	Source of net change (acres) in developed land			Percentage contribution to net change in developed land		
Land-use type	1986-1995	1995-2002	1986-2002	1986-1995	1995-2002	1986-2002
Upland forest	7,398	6,038	13,437	81.3	75.1	78.4
Upland agriculture	1,190	1,036	2,227	13.1	12.9	13
Wetlands	302	229	531	3.3	2.8	3.1
Barren land	222	703	925	2.4	8.7	5.4
Wetland agriculture	5	19	25	0.1	0.2	0.1
Water	-16	14	-2	-0.2	0.2	0
Total net change (acres)	9,102	8,040	17,142			

The ecological effect of forest fragmentation and changes in patch size associated with development in the Pinelands was addressed by a study conducted by Gibson et al. (1988) who compared 19 oak-pine forest fragments to 16 contiguous oak-pine forest stands. Forest fragments were defined as stands bordered by >50% agricultural or developed lands. Contiguous stands were defined as stands bordered by <50% agricultural or developed lands. The contiguous oak-pine stands were usually bordered by >75% upland pine-oak or pitch pine forest. Sapling density, diversity, and richness and tree-species richness were higher in the oak-pine fragments than in the contiguous forest stands. Species richness of forest fragments increased with area. The higher species richness in oak-pine-forest fragments was due to the large number of species that were unique to the fragments. Stoltzfus (1990) studied the effects of swamp size on the species composition and community structure of mature Atlantic white cedar swamps within forested Pinelands watersheds. He found that hydrology and past disturbance history were important determinants of swamp composition and structure and that fragmentation resulting from fire and timber harvesting had relatively little effect on these two characteristics.

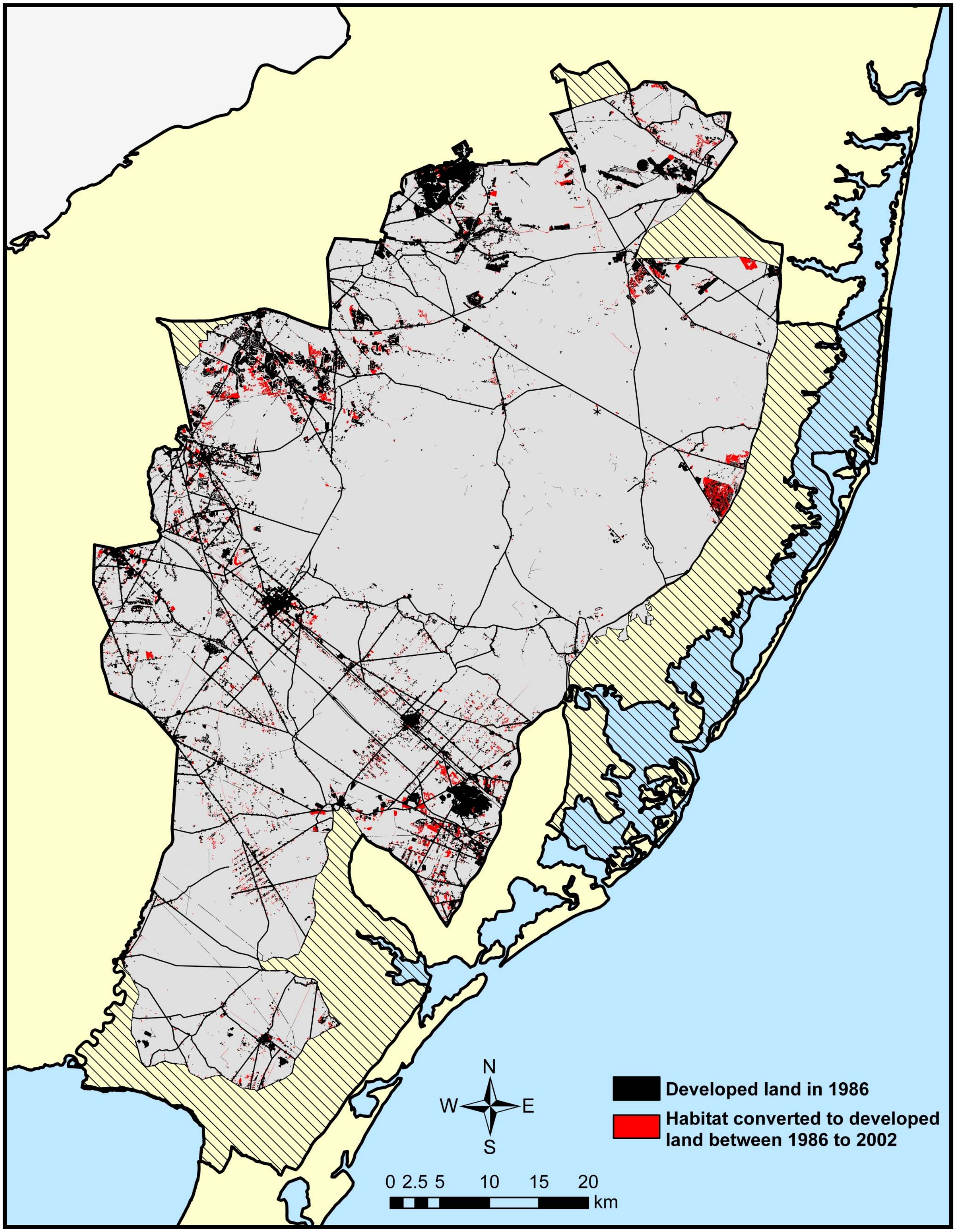

Figure 1.3. Pinelands habitat (upland forest, wetlands, and water) that was converted to developed land between 1986 and 2002. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

Edge

Habitat loss and fragmentation create edge, which is the portion of an ecosystem or habitat near its perimeter where environmental conditions differ from those in the interior (Forman 1995, Turner et al. 2001). The term is also used as a measure of the length of adjacency between cover types on a landscape. The ratio of edge to interior or core area increases with a decrease in patch size (Kennedy et al. 2003). Edges may be inherent or induced (Yahner 1988). Inherent edges are due to differences in natural environmental conditions such as soil or topography. Induced edges are created by natural or anthropogenic disturbance. Some natural phenomena resulting in induced edge are fire, wind, and insect outbreaks (Harper et al. 2005).

The edge-effect concept recognizes that species abundances and diversity may differ along edges compared to the interior of a habitat patch (Leopold 1933, Forman 1995, Sisk and Haddad 2002, Harper et al. 2005). Temperature, moisture levels, light intensity, wind, and nutrient cycling can all vary between the edge and the interior of a habitat patch (Saunders et al. 1991, Murcia 1995, Debinski and Holt 2000). Disturbances, such as fire, wind damage, and human activities, may occur at a higher frequency along edges compared to the interior (Kennedy et al. 2003, Murcia 1995).

Edge may increase overall wildlife diversity, but it can have negative consequences for wildlife (Yahner 1988). Most empirical evidence for the negative effects of fragmentation describes the habitat degradation that occurs in patches with biotic- and abiotic-edge effects (Harrison and Bruna 1999). Parker et al. (2005) concluded that studies reporting an area effect on forest birds are confounded or potentially confounded by edge effects. For example, nest predation and cowbird parasitism may be greater in fragmented landscapes (Brittingham and Temple 1983, Gates and Gysel 1978, Wilcove et al. 1986, Paton 1994, Robinson et al. 1995, Keyser et al. 1998, Batáry and Báldi 2004), although this effect is not always observed (Hanski et al. 1996a). Murcia (1995) suggested that because most studies are species- and site-specific, generalizations about edge effects cannot be made.

Created edge associated with timber harvesting, human-related wildfire, and historic resource exploitation is a prominent feature of the Pinelands landscape. In this human-dominated ecosystem where landscape patterns reflect the legacy of historic land uses, this type of edge may be viewed as a characteristic feature of the ecosystem that positively influences Pinelands biodiversity. As discussed in a subsequent section on land-use effects, created edges along roads, farmland, and developed areas and the activities associated with these land uses may have a far different and potentially negative effect on the ecosystem.

Minimum Viable Populations and Minimum Areas

Adapting Forman’s (1995) definition of patch, Turner et al. (2001) added that a patch is a continuous area of space with all necessary resources for the persistence of a local population that is separated from other patches by unsuitable habitat. Determining the minimum area required to maintain viable populations of all species comprising the regional-species pool requires an understanding of the relationship of population size to area and extinction probabilities (Shaffer and Samson 1985, Simberloff and Abele 1982). Just how large a habitat patch needs to be depends on the reproductive rate and population genetics of the organism in question, the rate of emigration from the patch, landscape patterns, matrix quality, and stochastic factors such as disturbances (Fahrig 2001, Soule and Simberloff 1986). A large amount of habitat may be needed for the persistence of species with low reproductive potential and a risky dispersal strategy (Fahrig 2001) or for populations in fragmented landscapes (Fahrig 2002).

Shaffer (1981) defined a minimum viable population for a specified species in any given habitat as “the smallest isolated population having a 99% chance of remaining extant for 1,000 years despite the foreseeable effects of demographic, environmental, and genetic stochasticity, and natural catastrophes.” More to the point, Shaffer and Samson (1985) asked “what size population occupying what area has the required probability of persistence?” Noting the tentative nature of the definition, Shaffer (1981) indicated that the probability of survival and time frame was a matter of discussion between conservationists, planners, and natural-resource managers.

Population-viability-analysis models offer the promise of providing a method to predict the probability that a population of a particular species, especially a rare species, will be above a minimum size some time in the future (Reed et al. 2002, Morris and Doak 2002). However, such models are limited by several factors including the scarcity of adequate data needed for accurate predictions and the difficulty of replicating and validating the models (Boyce 1992, Coulson et al. 2001, Morris and Doak 2002).

Two widely cited population-viability estimates that have conservation applications were developed for the grizzly bear (*Ursus arctos horribilis*) (Shaffer 1981, Shaffer and Samson 1985) and spotted owl (*Strix occidentalis caurina*) (Lande 1988). Reed et al. (2003) estimated the minimum-viable-population size for 102 vertebrate species, including several species found in the Pinelands, such as the tiger salamander and timber rattlesnake. Minimum-viable-population size was defined as one with a 99% probability of persistence for 40 generations, with a median generation length of 5 years. Based on a population-viability analysis, they concluded that the minimum-viable-population size did not differ significantly among species or with latitude or trophic level and that sufficient habitat capable of supporting approximately 7,000 breeding-age adults is required to maintain long-term minimum-viable populations of vertebrates in the wild. This estimate assumed that the individual populations were not distributed as metapopulations and that no habitat loss occurred over the given time frame. Reed et al. (2003) noted that much larger minimum populations would be needed if anthropogenic factors and habitat destruction were driving a species to extinction.

Regardless of whether minimum-viable-population size varies among species and regions or whether a single value (e.g., 7,000 adults) can be used for all vertebrates, a determination of the minimum area needed to support such a target-population size is still needed. Reed et al. (2003) indicated that few contiguous landscapes capable of supporting 7,000 large vertebrates, especially carnivores, exist and recommended that management of a network of smaller populations be given high priority, an opinion that was previously voiced by Noss et al. (1996). By comparing species-area relationships for disturbance-intolerant, terrestrial-mammal species (i.e., species that do not persist in human-dominated landscapes) in parks of eastern Canada to estimate species-area relationships that existed prior to European settlement, Gurd et al. (2001) estimated that a minimum reserve area in the range of 2,700-13,362 km^2 (1,042-5,159 mi^2) would be required to prevent the loss of terrestrial-mammal species within the Alleghenian-Illinoian mammal province, which includes the northeastern United States.

Several studies that may bear greater relevance to the Pinelands indicate that large tracts of land are required to protect a region’s avifauna and snake populations. Based on studies conducted in north-central Florida, Dodd and Barichvich (2007) suggested that large terrestrial snakes need large amounts of contiguous habitat to maintain populations. Attempts have been made to identify species-specific, minimum-area requirements for birds. Robbins (1980) and Robbins et al. (1989b) estimated minimum areas of contiguous forest required to sustain viable populations of area-sensitive forest birds based on the point at which population levels begin to decline. The range of minimum areas of contiguous forest given by Robbins (1980) ranged from 3 ha (7.4 acres) for the great crested flycatcher to 1,000 ha (2,470 acres) for the worm-eating warbler. Robbins et al.

(1989b) suggested that the minimum area needed to retain all species of forest-breeding birds found in the Middle Atlantic States is 3,000 ha (7,410 acres). Burke and Nol (2000) indicated that forests <500 ha (1,236 acres) represent habitat sinks for area-sensitive species such as the ovenbird, which is a species found in the Pinelands, and that conservation efforts should focus on the protection of large forest fragments. Although useful, such estimates should be viewed with the caveat that habitat area reported for individual bird species varies among different studies.

Based on simulation results involving a hypothetical organism in a hypothetical landscape, Fahrig (2001) suggested that the minimum amount of habitat needed for the persistence of all species in a region will vary among regions and that there is no single habitat target. She recommended that habitat preservation and restoration be the first priority for conservation and that the quality of the whole landscape, including the matrix, be improved by maintaining a diverse landscape structure.

Metapopulations

Movement of species across the landscape, which is determined by landscape connectivity (Tischendorf and Fahrig 2000), is important for the maintenance of small and isolated populations (Macdonald and Johnson 2001, Revilla et al. 2004). These isolated populations may represent a metapopulation, which is a set of local populations within some larger area, where migration from one local population to some other patches is possible (Hanski and Simberloff 1997). Because the populations that make up a metapopulation may undergo repeated extinction and recolonization in suitable habitat patches, long-term persistence of the metapopulation depends on a balance between local extinction and recolonization through dispersal (Revilla et al. 2004). As an apparent extension of Shaffer and Samson’s (1985) definition of minimum population size, Hanski et al. (1996b) defined the minimum-viable-metapopulation size as “the minimum number of interacting local populations necessary for long-term persistence of a metapopulation in a balance between local extinctions and recolonizations.” Where disturbance produces a mosaic of successional patches whose longevity may influence the extinction rate of some species, the smallest area with a natural disturbance regime that minimizes extinctions of small populations by maintaining internal-recolonization sources represents the “minimum dynamic area” (Pickett and Thompson 1978).

Although no empirical studies of Pinelands metapopulations have been conducted, timber rattlesnake metapopulations may exist in southeastern Pennsylvania (Bushar et al. 1998), and Martine et al. (2005) suggested that broom crowberry (*Corema conradii*) might occur as a metapopulation in the Pinelands. Pond-breeding amphibians, such as the Pine Barrens treefrog, may also exist as metapopulations, where subpopulations periodically undergo extinction and colonization, with the latter dependent on the spatial arrangement of ponds. However, Marsh and Trenham (2001) indicated that breeding-pond isolation might be an oversimplified explanation of population extinction since many amphibian species spend a majority of their time in terrestrial habitats and amphibian dispersal abilities may have been underestimated in many mark and recapture studies. Johnson and Semlitsch (2003) also noted that individuals of some amphibian species could move between adjacent breeding sites and that simple metapopulation models may not apply to all amphibian populations. Pond isolation effects may be more important in highly disturbed habitats where the distribution of terrestrial habitats may play a greater role in determining amphibian population persistence than breeding-pond characteristics (Marsh and Trenham 2001). Another major consideration is that what appears to be an extinction of a local amphibian population may only reflect normal ranges of fluctuation captured by short-term monitoring efforts (Blaustein et al. 1994). Such fluctuations have been observed in Pinelands ponds (Table 1.2, Figure 1.4).

Harrison (1991) found few empirical studies where the classical view of metapopulations as sets of local populations persisting in a balance between extinction and colonization was observed and concluded that the evidence suggests a diminished role for local extinction in metapopulations. The need for any spatially explicit metapopulation model to be highly species-specific due to varying dispersal capacities among different species also limits the applicability of the metapopulation concept (Knight and Landres 2002). Regardless, because the total amount of habitat in a landscape has been shown to be a good predictor of metapopulation persistence, simply preserving as much habitat as possible may be a prudent conservation approach (Hanski 1998).

Table 1.2. Frequency of occurrence (%) of 11 anuran species at 20 Mullica River Basin ponds in each of 11 years. Source: Pinelands Commission, unpublished data.

Species	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Northern cricket frog	5	0	0	0	0	0	0	0	0	0	0
Bullfrog	14	19	24	5	10	0	0	0	5	14	10
New Jersey chorus frog	29	5	5	24	14	10	5	19	10	10	5
Northern gray treefrog	33	0	14	0	10	5	14	24	14	10	24
Fowler's toad	43	0	19	29	19	10	14	24	29	29	24
Wood frog	33	10	48	38	29	14	19	43	33	48	57
Carpenter frog	57	43	52	43	48	43	29	29	38	43	38
Green frog	86	67	76	24	62	43	14	76	86	86	48
Southern leopard frog	76	86	86	67	81	95	48	81	90	86	81
Pine Barrens treefrog	86	86	86	76	86	90	76	90	90	81	86
Northern spring peeper	90	81	86	90	81	95	71	90	90	90	81

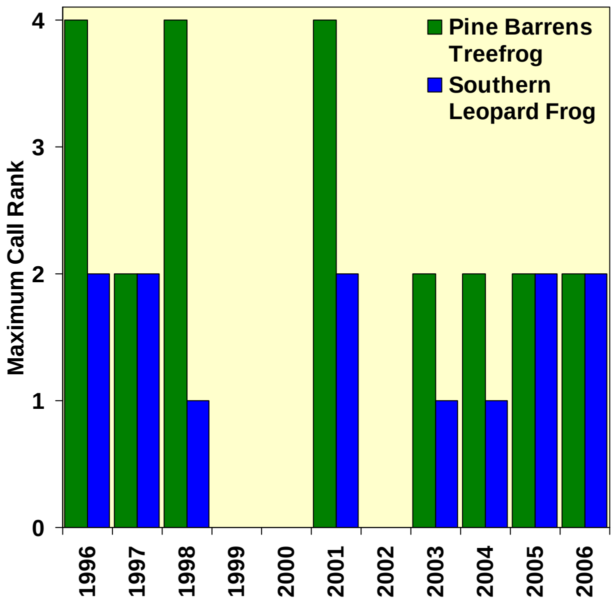

Figure 1.4. Variations in maximum call ranks for two frog species heard at a Mullica River Basin pond over an 11-year period. Call ranks represent the number of vocalizing males heard, where 0 = none, 1 = 1 individual, 2 = 2-5 individuals, 3 = 6-10 individuals, and 4 = >10 individuals calling. Source: Pinelands Commission, unpublished data.

Upland-wetland Mosaics

Studies conducted throughout the United State have emphasized the need to preserve a complex of upland and wetland habitats to sustain wetland-dependent amphibian and reptile populations. Buhlmann et al. (1993) found that some amphibian species in the Coastal Plain of Virginia live primarily in upland habitats and that other species that are typically associated with wetlands also occur in adjacent uplands. They also observed that most small mammals used both wetland and upland habitats. Bodie (2001) summarized nesting, overwintering, and migration movement data for 10 turtle species from the United States, Canada, and South America, and suggested that a riparian zone that extends 150 m (492 ft) from the stream edge is needed to encompass the majority of freshwater turtle migrations. Based on a summary of published data on terrestrial-habitat use by six species of pond-breeding ambystomid salamanders, Semlitsch (1998) recommended that a 164-m (538-ft) terrestrial-buffer zone be established from the edge of the aquatic habitat. Semlitsch and Jensen (2001) referred to this buffer zone as core-terrestrial habitat.

Pursuing the idea of a core-terrestrial habitat, Semlitsch and Bodie (2003) summarized data from the literature on the use of terrestrial habitats by 32 amphibian species and 33 reptile species associated with aquatic habitats. The mean core-terrestrial habitat ranged from 159 to 290 m (522 to 951 ft) from the edge of the associated aquatic habitat for amphibians and from 127 to 289 m (417 to 948 ft) from the edge of the associated aquatic habitat for reptiles. Johnson and Semlitsch (2003) suggested that for the protection of local gray treefrog populations, a minimum core-terrestrial habitat of 60 m (197 ft) should be established around breeding sites for activities occurring within the breeding season (non-breeding terrestrial habitat was not characterized). Crawford and Semlitsch (2007) recommended that a core-terrestrial habitat for stream-breeding salamanders be surrounded by an additional 50 m (164 ft) of habitat to buffer edge effects. Baldwin et al. (2006) indicated that the core-habitat approach might also conserve nonessential habitat for pool-breeding amphibians. They suggested using a species-specific approach, where breeding pools are linked with discrete habitats, such as upland-wintering areas, within known maximum-migratory distances for the particular species. One problem with this approach is that detailed information on animal movements is rarely known and results based on individual studies may not be applied to other populations or landscapes.

Burke and Gibbons (1995) radio-tracked 93 gravid turtles, including 73 mud turtles (*Kinosternon subrubrum*), a species that is native to the Pinelands, around a semipermanent Carolina bay wetland in South Carolina. Twenty-four mud turtles were also radio-tracked later in the season to locate their hibernation burrows. Ninety percent of the turtle nests and hibernation sites were found within a 73-m (239-ft) buffer surrounding the wetland. A 275-m (902-ft) buffer encompassed 100% of all nests and hibernation sites.

Madison and Farrand (1998) tracked 27 eastern tiger salamanders in and around four ponds surrounded by pitch pine/oak forests in the Long Island Pine Barrens. After leaving their pond, 23 salamanders moved in all directions within wooded areas, but avoided paved roads, commercial areas, and grassy fields. In New Jersey, eastern tiger salamanders primarily are found in Atlantic, Cape May, and Cumberland counties where this endangered species typically breeds in ponds and may move into the uplands following the breeding season (Liguori and Clark 2003). The general absence of tiger salamander in the central Pinelands may be due to its sensitivity to low pH since the species generally fails to reproduce in waters where the pH is less than 4.5 (Freda and Morin 1984). Thus, protection of a complex of upland and wetland habitats rather than acid-water conditions is probably more important where this species is found in the region.

The need to protect lands surrounding eastern tiger salamander breeding pools is supported by a study of a related species, the California tiger salamander (*Ambystoma californiense*). Trenham and Shaffer (2005) trapped adult and subadult California tiger salamanders in the area surrounding an isolated breeding pond (a playa vernal pool). Captures of adults declined with distance from the breeding pond. Captures of subadults increased steadily from 10 to 400 m (33 to 1,312 ft) from the pond and declined to zero at 800 m (2,625 ft). By relating captures to the distance from the pond, Trenham and Shaffer (2005) estimated that upland habitat extending up to 620 m and 630 m (2,034 ft and 2,067 ft) from the edge of the pond would be required to encompass 95% of the adults and subadults, respectively.

In the Pinelands, Pine Barrens treefrog breeding ponds are found in both upland and wetland landscapes (Bunnell and Zampella 1999, Zampella and Laidig 2003). Using radioactive tags, Freda and Gonzalez (1986) tracked the summer movements of eight Pine Barrens treefrogs captured at a seepage stream surrounded by a narrow shrub wetland in the New Jersey Pine Plains. Seven individuals remained within 70 m (230 ft) of the breeding site and one frog moved 106 m (348 ft) from the pond. Habitat descriptions given by Freda and Gonzalez (1986) suggested that most of the recaptures occurred in the surrounding upland habitats.

Although the need to preserve uplands to sustain wetland-dependent species is most often emphasized, many animal species associated with upland habitats also depend on wetlands. In the central Pinelands, timber rattlesnakes rely primarily on upland forests for summer foraging but hibernate in wetlands (Reinhart and Zappalorti 1988a, 1988b). Individual rattlesnakes move over large areas, with reported home ranges of 1.5 ha (3.7 acres) to 722 ha (1,784 acres) (Laidig and Golden 2004, Reinert and Zappalorti 1988). On average, nine timber rattlesnakes radio-tracked by Laidig and Golden (2004) for complete active seasons used upland pine and oak forests about 70% of the time and wetlands about 30% of the time.

The need to preserve a mosaic of different habitat types is also suggested by the results of a Pinelands breeding-bird survey conducted by Laidig (1997) along upland-wetland ecotones in the Mullica River Basin. He reported that all 24 common species detected in the survey were found in more than one habitat type, and all but two were found in both uplands and wetlands. Wander (1980) surveyed breeding birds in Pinelands cedar swamps and found that the ecotones between cedar swamps and adjacent habitats supported a higher diversity of birds and that birds moved among and between habitats. Brush (1987) found that most birds within his central Pinelands study area switched between and among upland and lowland habitats seasonally. He concluded that no one habitat was sufficient to allow all species to coexist and that a mosaic of contiguous vegetation types must be preserved to maintain the full diversity of Pinelands birdlife. Kerlinger (1984) also recommended that to maintain a characteristic Pinelands avifauna, a mosaic of successional habitats that reflect the region's dynamic-disturbance regimes must be maintained.

The barred owl (*Strix varia*) is a wide-ranging species that occurs in a variety of woodland habitats. In Minnesota, Nicholls and Warner (1972) found that barred owls displayed a preference for oak woods and mixed hardwoods and conifers compared to northern white cedar (*Thuja occidentalis*) swamps, oak (*Quercus* spp.) savannas, alder (*Alnus* spp.) swamps, marshes, and old fields. Survey work in southern New Jersey has elicited vocal responses from barred owls in oak-pine uplands, hardwood swamps, pitch pine lowlands, and Atlantic white cedar swamps (Sutton 1988, Laidig and Dobkin 1995), indicating that the species ranges across upland and wetland landscapes.

Small-mammal species are also found across a range of upland and wetland Pinelands habitats (Connor 1953, White 1961, Craig and Dobkin 1993), which indicates that protecting a range of habitat types rather than individual habitats may also benefit small-mammal biodiversity. For example, Connor (1953) trapped a total of ten small-mammal species in ten different habitats in the central Pinelands of Ocean and Burlington counties, with most species found in both upland and wetland habitats or the ecotone between these habitats.

The Effect of Land Use on Species Richness, Diversity, and Abundance

Birds. Land uses can have a significant effect on regional avifauna. Dunford and Freemark (2005) indicated that the effects of urban- and agricultural-land uses on forest birds can vary between species, occur at different spatial scales, and be positive or negative, with Neotropical and short-distance migrants being the most sensitive to human-dominated landscapes. Many early habitat-island studies showing that habitat area influenced bird species composition and richness were conducted in forest fragments embedded in what were primarily agricultural landscapes (Martin 1981, Galli et al. 1976, Forman et al. 1976, Blake and Karr 1984, Freemark and Merriam 1986). More recently, McIntyre (1995) surveyed birds in a Georgia Piedmont agricultural landscape and found that the mean number of bird species per patch per sampling date was higher in contiguous forest patches compared to small (< 3.25 ha or < 8 acres) and large (10-13.25 ha or 24.7-32.7 acres) forest fragments, but overall total species richness was higher in the isolated patches. The total and mean number of interior species was higher and the total and mean number of edge species was lower in the contiguous forest patches. Best et al. (1995) reported that the total numbers of bird species in agricultural landscapes of Iowa were higher in lowland deciduous forests adjacent to streams (floodplain forests), deciduous upland forests (upland forests), and shrub land compared to a range of agricultural-land uses.

A majority of bird species observed during a survey of Wisconsin cranberry beds and adjacent unmodified wetlands were absent or infrequently observed in the cranberry beds (Jorgensen and Nauman 1993a). In another Wisconsin cranberry-wetland-system survey, fifteen species of birds, reptiles, amphibians, and mammals were observed in or foraging over cranberry beds, whereas the number of species associated with reservoirs, ditches/levees, and other disturbed areas ranged from 21 to 30 (IEP 1990). A total of 19 and 30 species were observed in adjacent uplands and wetlands, respectively. Although fewer species may be associated with cranberry beds compared to other habitats, impoundments associated with commercial cranberry operations can be important for migrating waterfowl (Jorgensen and Nauman 1993b).

Mixed urban-agricultural landscapes can also influence the composition of bird communities. Croonquist and Brooks (1993) reported that bird-species richness and abundance decreased with distance from a stream in a Pennsylvania watershed with more than 29% altered land (agriculture and residential land), whereas almost no change was observed in another watershed where undisturbed forest covered 94% of the area. Dunford and Freemark (2005) emphasized that urban and agricultural land in the surrounding matrix can affect forest-bird abundance and species richness. They found that total species abundance and the abundance of short-distance migrants decreased as the percentage of intense (row-crop) agriculture within 5,000 m (16,404 ft) of a survey point increased. The number of Neotropical migrant species decreased as the percentage of urban land within both narrow (within a surrounding 1,800-m or 5,905-ft radius) and broad (within a surrounding 5,000-m radius) spatial scales increased. Neotropical-migrant species abundance also decreased as the percentage of urban land within 1,800-m increased. Resident-species richness was higher at sites surrounded by both row-crop agriculture and urban land within the broader scale. Edge-species abundance was also higher at sites surrounded by row-crop agriculture within the broader scale. Both edge-species abundance and resident abundance decreased as the percentage of low-intensity agriculture (pasture, hay fields, old fields) within the broad-scale landscape increased.

Residential development frequently results in a decrease in Neotropical and forest-interior birds and an increase in birds typically associated with urban areas. Whitcomb (1977) suggested that human impacts associated with suburbanization, such as trampling, can have a greater effect on Neotropical birds than on other species because many species nest on or near the ground. Suburbanization can also directly affect predation. Wilcove (1985) found that predation of artificial nests was more intense in woodlots surrounded by residential development compared to similar woodlots surrounded by agricultural land. He attributed the higher predation to higher densities of nest predators such as blue jay (*Cyanocitta cristata*), raccoon (*Procyon lotor*), dogs (*Canis familiaris*), and house cats (*Felis domesticus*) near suburban developments. An increase in nest predation is not limited to urban areas. Bayne and Hobson (1997) reported that artificial ground nests placed in forested patches in a Canadian agricultural landscape had significantly higher rates of predation than nests in either logged or contiguous forest landscapes.

Butcher et al. (1981) compared the results of breeding-bird surveys conducted from 1953 to 1976 in a 23 ha (57 acres) Connecticut woodland located adjacent to an area that had become progressively suburbanized. Although species diversity remained high, densities of forest birds declined quite precipitously and species typical of suburban areas increased in number during the survey period. Butcher et al. (1981) concluded that in addition to destroying forest habitat and isolating the study site from similar forest habitat, development reduced the buffer of low-density human use, creating disturbance from construction, noise, lights, and other human activities. A subsequent study of the same Connecticut woodland found that although suburban species continued to increase after 1976, some long-distance migrants that had previously declined increased in abundance (Askins and Philbrick 1987). This increase was attributed to reforestation of abandoned farmland in surrounding areas. The abundance of forest-dwelling long-distance migrants was negatively correlated with suburban-bird abundances.

Kluza et al. (2000) observed that Neotropical migrants and forest-interior-bird species were less abundant in Massachusetts forest sites associated with moderate-housing densities compared to forest sites associated with low-housing densities. Neotropical migrants and forest-interior species were less abundant and blue jays were more abundant in moderate-housing-density forests. Kluza et al. (2000) suggested that nest predators might be responsible for the reduced densities in the forest with moderate-housing densities. Friesen et al. (1995) also examined the effect of forest size and housing densities on Neotropical-migrant diversity and abundance. Both diversity and abundance increased as forest size increased, but decreased dramatically as the number of adjacent houses increased, regardless of forest size.

Beissinger and Osborne (1982), who compared bird communities in a residential area and an undisturbed climax beech-maple forest, associated urbanization with a decrease in species richness and diversity and dominance by a few species. Both bird-species richness and diversity in isolated Massachusetts woodlands increased with woodland size and decreased with increasing building density in surrounding urban lands (Tilghman 1987). In the Santa Clara Valley of California, bird species richness in riparian woodlands decreased in relation to the proximity of buildings and the number of nearby bridges (Rottenborn 1999). Smith and Wachob (2006) also reported that breeding-bird-species richness and diversity in riparian habitats along the Snake River in Wyoming decreased with increasing residential development. Development had a disproportionate effect on Neotropical migrants, which became less common as the exotic European starling (*Sturnus vulgaris*) increased in abundance.

The presence of exotic species, such as the European starling, and an increase in the number of bird species typical of urban areas are generally associated with suburbanization. In Seattle, Washington, Gavareski (1976) observed a marked increase in the number of species typical of urban areas in small highly modified parks compared to large forested parks with native vegetation. The American robin (*Turdus migratorius*), northern mockingbird (*Mimus polyglottos*), and house sparrow (*Passer domesticus*) were especially abundant in the small (1-5 ha or 2.47-12.4 acre) woodlands surrounded by urban lands in Massachusetts (Tilghman 1987). The American robin, brown-headed cowbird, and house wren (*Troglodytes aedon*) were among six bird species that decreased in density with the distance from houses in a Colorado exurban development

(development found in a matrix of native habitat) (Odell and Knight 2001). The same three species, along with the European starling, were found at higher densities in native habitat within the developments compared to undeveloped areas. In another study of exurban development in Colorado, compared to undeveloped areas, grasslands surrounding both the clustered and dispersed housing developments had higher densities of nonnative birds and bird species that frequently live in association with humans, including the European starling, common grackle, American robin, mourning dove, and rock dove (Lenth et al. 2006).

In Virginia, Aldrich and Coffin (1980) found a larger total breeding-bird population and a greater number of bird species following suburbanization, but several Neotropical species that were once present were absent. The increase in species richness was due to the establishment of suburban birds such as blue jay, northern mockingbird, European starling, northern cardinal (*Cardinalis cardinalis*), and song sparrow (*Melospiza melodia)*. The gray catbird (*Dumetella carolinensis*), American robin, and house sparrow, which were absent before suburbanization, were also numerous after suburbanization occurred. Aldrich and Coffin (1980) described all eight species as common suburban-residential breeding birds in other regions of North America, including New Jersey. American robin, European starling, chipping sparrow (*Spizella passerina*), rock dove (*Columba livia*), and house sparrow are common near developed areas in the Pinelands (Leck 1979). With the exception of the chipping sparrow, blue jay, and American robin, which are widely distributed in the Pinelands (Walsh et al. 1999), breeding populations of the birds associated with developed areas by Aldrich and Coffin (1980) and Leck (1979) are generally absent in the core of the Pinelands (Walsh et al. 1999).

In California, bird-species composition along a rural-to-urban gradient that ranged from undisturbed oak woodland to a business district, shifted from a community dominated mostly by native species to one with invasive and exotic species (Blair 1996). In Ontario, Canada, the density and percentage of wintering alien birds in urban areas with significant areas of remnant natural vegetation increased with the percentage of urban land in a surrounding 1-km (0.62-mi) buffer (Smith 2007).

In a comparative study of New York (Long Island and Albany) and New Jersey upland Pine Barrens habitats, Kerlinger and Doremus (1981) associated the decline of four characteristic Pine Barrens bird species, including ovenbird, brown thrasher (*Toxostoma rufum*), pine warbler (*Dendroica pinus*), and prairie warbler (*D. discolor*), in the New York barrens with edge effects created by a reduction in habitat size and dissection associated with roads and developments. They identified cowbird parasitism, nest predators, and fire suppression as other factors that can affect bird-community structure.

Habitat loss and fragmentation of a forest landscape associated with suburbanization may be an especially important concern for some raptors such as the barred owl and red-shouldered hawk (*Buteo lineatus*). A study conducted in northern New Jersey revealed that compared to other owl species, barred owls were more frequently found in areas with no clearings or trails and less frequently found in areas with clearings or near human habitation (Bosakowski et al. 1987). In a second northern New Jersey study (Bosakowski 1994), nearly all of the barred owl occurrences were in forest, whereas occurrences in agricultural and low-density urban areas were much lower than expected.

Breeding red-shouldered hawks in New Jersey are primarily limited to the deciduous lowland swamp forests in the far south (Dowdell and Sutton 1993) and moist lowlands in the north (Leck 1984). Dowdell and Sutton (1993) suggested that increased forest fragmentation in southern New Jersey may lead to the replacement of red-shouldered hawks by red-tailed hawks (*Buteo jamaicensis*) and increased predation pressure on the red-shouldered hawks from great horned owls

(*Bubo virginianus*). However, a study conducted in Ohio (Dykstra et al. 2000) found that reproductive rates of red-shouldered hawk were similar at rural-nesting and suburban-nesting sites.

Extensive forest has also been associated with Cooper's hawk (*Accipter cooperii*) territories in New Jersey (Bosakowski et al. 1992a), and the species has been described as being intolerant of human disturbances such as suburban development (Bosakowski et al. 1993). However, conflicting accounts have been given regarding the need for large tracts of forest by the Cooper's hawk and its tolerance of agricultural or suburban and urban development in New Jersey (Liguori 2003). Cooper's hawk can successfully nest in urban areas (Boal and Mannan 1998), although urban environments may function as ecological traps due to nestling-mortality associated with diseases such as trichomoniasis (*Trichomonas gallinae*) found in urban birds (Boal and Mannan 1999). In the New Jersey-New York Highlands, Cooper's hawks nested closer to development and roads and were found to be more tolerant of automobile traffic than goshawks (Bosakowski 1992b), a characteristic also described by Bosakowski (1992a).

Mammals. The effect of urban and agricultural land on mammals is variable. Crooks (2002) examined the distribution of mammalian carnivores in small urban-habitat fragments and less disturbed areas in southern California. Although the landscape consisted primarily of dendritic canyons dissecting coastal mesas and is unlike that of the Pinelands, several species studied are also found in New Jersey. The probability of occurrence of long-tailed weasels (*Mustela frenata*), bobcats (*Lynx rufus*), and coyotes (*Canis latrans*) was lower in smaller habitat patches, whereas the probability of encountering striped skunks (*Mephitis mephitis*), raccoons, gray foxes (*Urocyon cinereoargenteus*), and opossums (*Didelphis marsupialis*) was not related to habitat-patch area. Another study conducted in southern California found that native rodents became locally extinct in habitats fragmented by urbanization and that fragments supported fewer species than unfragmented habitats (Bolger et al. 1997). In Colorado, dogs and house cats were detected more frequently near houses in a Colorado exurban development, whereas the opposite trend was observed for red foxes and coyotes (Odell and Knight 2001). With the exception of dogs, Lenth et al. (2006) found no significant difference in the presence of individual mammal species between clustered exurban development, dispersed exurban developments, and undeveloped areas in Colorado. In southeastern Wisconsin, Matthiae and Stearns (1981) found that the mammalian community changed in composition and abundance from rural to urban areas, with more diverse communities found at rural sites. Forest islands located along the transition between rural to urban areas had lower mammalian-species richness and abundance compared to forest islands in rural and urban areas.

Bayne and Hobson (1998) compared the abundance of small mammals in contiguous forest, forest patches surrounded by logged forest, and farm woodlots in Saskatchewan, Canada. The total abundance of all small mammals was lowest in forest patches isolated by logging, due primarily to lower deer mouse abundance. The red squirrel (*Tamiasciurus hudsonicus*), a species also found in the Pinelands, was more abundant in farm woodlots. A comparison of the number of small mammals captured on Wisconsin cranberry beds and in adjacent habitats within 50 m (164 ft) of the beds and at distances greater than 100 m (328 ft) from the beds revealed that small mammals were present in greater numbers in the adjacent habitats (Jorgensen and Nauman 1995).

In northwestern Ohio, white-footed mice were present in and moved among five different habitat types, including woods, the edge of woods, roadside ditches, crop fields, and farmsteads (Cummings and Vessey 1994). Barko et al. (2003) captured more white-footed mice in bottomland forest patches surrounded by urban land than in patches surrounded by upland deciduous forest. Heske (1995) found no difference in furbearer activity and small-mammal abundance between forest-farm edge and forest-interior sites in southern Illinois. Similarly, Rosenblatt et al. (1999) concluded that the distribution of most mammal species surveyed in forest fragments in an Illinois agricultural landscape was not altered by the presence of agriculture.

No Pinelands mammal species is considered threatened or endangered (Beans and Niles 2003), although the status and distribution of most species in the region are not well documented and no studies have assessed the abundance and distribution of Pinelands mammals in relation to urban- or agricultural-land uses. Some mammals found in the Pinelands, such as the raccoon, eastern gray squirrel (*Sciurus carolinensis*), opossum, whitetail deer (*Odocoileus virginianus*), and striped skunk may coexist with and sometimes benefit from suburban development (Hoffmann and Gottschang 1977, Sexton 1990, Broadfoot 2001, DeStefano and DeGraaf 2003, Prange and Gehrt 2004). Connor (1953) indicated that neither skunks nor opossums were common in the central Pinelands.

As indicated by the location of dams, the beaver (*Castor canadensis*) is currently widespread in both undisturbed and heavily farmed and developed areas of the Mullica River Basin (Laidig and Sulikowski 2001). The black bear (*Ursus americanus*), which was extirpated in the Pinelands, has expanded its range from an initial stronghold in northwestern New Jersey to suitable habitat in the majority of New Jersey counties (NJDEP 2004). Sightings have been reported for all Pinelands counties (McConnell et al. 1997, NJDEP 2004, P. C. Carr, personal communication). In northern New Jersey, black bears are able to persist in habitats fragmented by residential development (Fimbel et al. 1991). Their ability to live in close proximity to humans has created human-bear conflicts that have increased with the increasing pace of development in northwestern New Jersey counties (NJDEP 2004).

Reptiles and amphibians. Numerous studies have documented that surrounding landscape characteristics, including the presence of forest, urban, and agricultural land, influence the distribution and abundance of wetland-dependent reptile and amphibian species, including many species that are found in the Pinelands. The effect of surrounding forest is generally positive, whereas that of urban and agricultural land is generally negative.

The presence of wood frogs and pond-breeding salamanders, including the spotted, Jefferson, and small-mouthed salamander (*Ambystoma maculatum*, *A. jeffersonianum*, and *A. texanum*), and overall salamander diversity in an intensely farmed Ohio landscape dominated by row-crop agriculture was positively associated with the percentage of forest within 200 m (656 ft) of the breeding ponds (Porej et al. 2004). In contrast, Pearl et al. (2005) reported that in Oregon's Williamette Valley, where 43% of the area is in agricultural use and 11% is urban and rural development, breeding populations of only one of five western amphibian species (none found in the Pinelands) was positively associated with the percentage of forest in the surrounding landscape. Rothermel and Semlitsch (2006) found that survival of juvenile spotted and marbled (*A. opacum)* salamanders was lower in experimental enclosures placed in old fields than in enclosures placed inside forest edge and within interior forest. In the same Missouri study area, both adult and recently metamorphosed-juvenile spotted salamanders were observed avoiding planted-grassland habitat that partially surrounded a breeding pond and showed a directional bias for adjacent-forest habitat (Rittenhouse and Semlitsch 2006). Agricultural activities can also directly impact reptile and amphibian populations. Saumure et al. (2007) attributed wood turtle (*Glyptemys insculpta*) death and injuries to agricultural machinery.

In a Maine landscape that was approximately 50% agriculture and 50% forested, Guerry and Hunter (2002) found that the probability of occurrence of wood frogs, green frogs, eastern newts (*Notopthalmus viridescens*), spotted salamanders, and blue-spotted/Jefferson salamanders (*Ambystoma laterale/A. jeffersonianum*) was positively associated with the amount of forest within a 1-km radius of a breeding pond, the occurrence of American toads and northern leopard frogs (*Rana pipiens*) was negatively associated with forest area, and the presence of spring peeper (*Pseudacris c. crucifer*) and mink frog (*Rana septentrionalis*) was not associated with the amount of forest surrounding a pond.

Amphibian-species richness in the urbanizing deciduous-forest ecoregion of Minnesota was reduced at sites with moderate to high proportions of urban land within 500 m (1,640 ft), 1,000 m (3,281 ft), and 2,500 m (8,202 ft) (Lehtinen et al. 1999). Species richness from both the deciduous-forest and agricultural-prairie ecoregions was negatively associated with the density of roads and the distance to the nearest wetland at all three spatial scales. A decrease in amphibian richness was also associated with urbanization in the Puget Sound region of Washington (Azous and Richter 1995, Richter and Azous 1995, Reinelt et al. 1998, Reinelt and Taylor 2001). Species richness of pond-breeding-amphibian species in a Canadian agricultural landscape was negatively correlated with the distance to the nearest woods and positively correlated with the percentage of woods within a 2-km (1.2 mi) radius (Hecnar and McCloskey 1998).

Gibbs (1998a) studied the distribution of amphibians along the length of a forest-fragmentation gradient in southern Connecticut where forest cover ranged from 5% at the urban end to 95% at the rural end. Redback salamanders and northern spring peepers were found in available habitat and appeared relatively resistant to fragmentation, wood frogs and spotted salamanders were absent where forest cover was less than about 30%, and red-spotted newts (*Notopthalmus v. viridescens*) were absent below a forest cover of about 50%.

Knutson et al. (1999) related the relative abundance and species richness for all anuran species and individual guilds (breeding, non-breeding, and hibernating assemblages) to landscape features in Wisconsin and Illinois. In both states, most measures were negatively associated with the presence of urban land within 1,000 m (3,281 ft) of the anuran-survey points. Relative abundance was positively associated with agricultural area in Wisconsin, but not in Illinois. Knutson et al. (2004) compared amphibian assemblages found in constructed-agricultural ponds and natural ponds in a southeastern Minnesota agricultural landscape. They found that amphibian-species richness and reproductive success in small constructed ponds surrounded by row crops was similar to that of natural ponds and ponds surrounded by ungrazed pasture. Both amphibian-species attributes were more closely associated with pond characteristics such as water quality, vegetation, and the presence of predators than with characteristics of the surrounding landscape. Turbidity and concentrations of total nitrogen and phosphorus were elevated in constructed ponds located adjacent to both grazed land and row-crop agriculture compared to natural ponds and constructed ponds adjacent to ungrazed land. Five of 10 species identified in the study are found in the Pinelands. Spring peeper and green frog are widespread in the Pinelands and throughout New Jersey. The eastern tiger salamander, northern gray treefrog, and pickerel frog (*Rana palustris*), are border-entrant species. Border-entrant species are anurans that are usually found in the Pinelands at sites disturbed by human activities (Conant 1979), and their absence from undisturbed areas has been attributed to the low pH of Pinelands waters (Gosner and Black 1957, Freda and Dunson 1986, Zampella and Bunnell 2000, Zampella et al. 2001).

Rubbo and Kiesecker (2005) reported that larval-amphibian-species richness in central Pennsylvania wetlands was lower in urban areas compared to rural areas, which they attributed to a decrease in the occurrence of wood frogs, spotted salamanders, and Jefferson salamanders. Other species, including the pickerel frog, spring peeper, northern gray treefrog, green frog, and bullfrog (*Rana catesbeiana*), were described as resilient to the effects of urbanization. Although the study area included urban and agricultural land, upland forest, and wetlands, Rubbo and Kiesecker (2005) included only the percentage of upland forest and wetlands within a 1-km (0.62-mi) radius of a survey site in their analysis.

Gibbs et al. (2005) evaluated changes in anuran populations over a 30-year period in the northern and western-central regions of New York. Developed lands or lawns surrounded sites where American toad and spring peeper populations were less persistent. With the exception of spring peeper, which declined in areas with row crops, there was little evidence of a negative association between population transitions and the extent of agriculture. Gibbs et al. (2005) attributed this to the low-intensity agriculture that characterizes the region studied.

Amphibian-species richness in a heavily forested region of New Hampshire was lowest in wetlands surrounded by less than 40% forest cover within a 1,000-m (3,281-ft) radius (Herrmann et al. 2005). The proportion of forest in 0 to 100-m (0 to 328-ft) and 1,000 to 2,000-m (3,281 to 6,562-ft) buffers surrounding the breeding sites had little influence on overall richness or the density of several species. Based on the results of their study, Herrmann et al. (2005) suggested that ponds surrounded by more than 60% forest within a 1,000-m radius might be needed to support species-rich amphibian assemblages. Houlahan and Findlay (2003) found that both amphibian-species richness and abundance in 74 Ontario wetlands was positively correlated with forest cover and negatively correlated with road density within 3,000 m (9,842 ft). They emphasized the need to maintain a heterogeneous regional landscape containing relatively large areas of natural forest and wetlands.

In the Pinelands, the distribution of Pinelands frogs and toads has also been associated with landscape patterns. Bunnell and Zampella (1999) compared the composition of anuran assemblages in 14 acid-water intermittent ponds located in forested landscapes along the northwestern boundary of the Mullica River Basin. The proximity of developed land and upland agriculture varied among the ponds. Ten species were found at the 14 ponds, including six native species and four border-entrant species. The native species heard in the ponds were the carpenter frog (*Rana virgatipes*), Pine Barrens treefrog, spring peeper, green frog, southern leopard frog (*Rana utricularia*), and Fowler’s toad (*Bufo woodhousii fowleri*). The border-entrant species, which included the cricket frog (*Acris c. crepitans*), northern gray treefrog, New Jersey chorus frog (*Pseudacris triseriata kalmi*), and wood frog, were heard vocalizing only at ponds located near landscapes altered by development and upland agriculture. Larvae of border-entrant species were not found in any of the ponds. Based on the results of the study, Bunnell and Zampella (1999) suggested that the distribution of adult anurans is influenced by landscape patterns, whereas larval recruitment may be limited by pond chemistry.

Zampella and Bunnell (2000) also reported an association between the composition of frog and toad assemblages and landscape characteristics during a vocalization survey conducted throughout the Mullica River Basin at sites representing a wide range of habitats, including natural- and excavated-intermittent ponds, abandoned-cranberry bogs, streams, and stream impoundments. Although most native Pinelands species were widely distributed, border-entrant species, including the northern gray treefrog, bullfrog, pickerel frog, and northern cricket frog, were found only at sites located in the western portion of the Mullica River where development and upland agriculture were more extensive. The study also revealed that the Pine Barrens treefrog and carpenter frog were generally absent and the leopard frog was less frequently heard where bullfrogs were found. Introduced bullfrogs have been associated with the decline of some native-frog species in other regions of North America (Moyle 1973, Hammerson 1982, Hayes and Jennings 1986). The association between border-entrant species and more highly altered landscapes, where surface waters are characterized by elevated pH and dissolved solids, and the negative relationship between carpenter frogs and bullfrogs has been found throughout the Pinelands (Zampella et al. 2001, 2003, 2005, 2006a)

Conservation of an endangered species such as the southern gray treefrog *(Hyla chrysoscelis*) highlights the difference between an approach that emphasizes individual species and one that emphasizes whole communities, ecosystems, and landscapes. Conant (1979) described the southern gray treefrog as a peripheral species that is restricted to the perimeter of the Pine Barrens and is not present in the region. He noted that it might eventually be found within the Pine Barrens. Although the historical range of the southern gray treefrog in New Jersey was limited to Cape May and

Cumberland counties, since the early to mid 1990's it has been found in southern Atlantic County and eastern Ocean County (Liguori 2003). It is possible that the range extension of the southern gray treefrog is associated with habitat degradation that accompanies development of coastal areas, since in eastern Ocean County the species has been found in stormwater basins located adjacent to the Stafford Township landfill (Zappalorti et al. 2006), in a wetland along Route 9, and in a residential stormwater basin along Route 72 (J. F. Bunnell, personal observation). Prior to being developed, the Ocean County area represented characteristic Pine Barrens habitat. In a survey of southern gray treefrogs in Cumberland, Cape May, and Atlantic counties, Zappalorti and Dowdell (1991) reported that 33% of 80 confirmed localities were borrow pits and 6% were man-made retention/detention basins. Like the northern gray treefrog and the bullfrog, the southern gray treefrog may be an indicator of altered-Pinelands-habitat conditions and reduced ecological integrity. Directing conservation efforts towards this species in some areas of the Pinelands may not be consistent with the ecological objective of preserving regional biodiversity.

Insects. Although development and upland agriculture may have a negative effect on some species, Pinelands bee-species richness and abundance respond positively to these land-uses. Winfree et al. (2007) surveyed bees at sites representing a range of Pinelands landscape settings and collected a total of 130 species. With the exception of three naturalized-exotic species, all were considered native to the Pinelands study area. Although 18 species were positively associated with extensive forest, bee abundance and species richness within forest habitats decreased as surrounding forest cover increased, whereas both metrics were positively associated with the percentage of agriculture in the surrounding landscape. Species richness was also positively associated with the extent of surrounding suburban land (low-density housing). High-density urban land was not related to either richness or abundance. At the local scale, agricultural fields and suburban and urban land had higher species richness and abundance than extensive forest, with the highest values associated with agricultural fields.

Plants. Several studies conducted in the Pinelands and other regions have indicated that land-use can influence the structure and species composition of both upland and wetland plant communities. Loeb (2006) reported that increasing human population is associated with a decrease in native-plant species and an increase in nonnative species in large urban parks along the northeast corridor between Washington, D.C. and Boston, Massachusetts. In a Canadian agricultural landscape, woodlot sites had higher total and native-plant-species richness compared to nine other site types, including crop, plantation, herbaceous-fencerow, and old-field habitats (Freemark et al. 2002b). In Colorado, the percentage cover of native-plant species was lower and the percentage cover of nonnative-plant species was higher in both clustered and dispersed exurban housing developments compared to undeveloped areas (Lenth et al. 2006). The number of nonnative species was similar regardless of housing-development pattern.

Burton et al. (2005), who studied riparian-forest vegetation in the southeastern United States, found that the proportion of nonnative-woody species increased and regeneration-layer-species diversity decreased along a rural- to urban-watershed gradient associated with proximity to an urban center. In a study of Canadian wetlands conducted by Houlahan et al. (2006), plant-species richness was positively correlated with wetland area and the amount of adjacent forest. Exotic-plant species were more likely to be found in wetlands without much surrounding forest. Similarly, species richness of submerged vegetation in Ontario marshes decreased along an environmental gradient characterized by an increase in the percentage of agricultural land and a decrease in percentage of forest land within a watershed (Crosbie and Chow-Fraser 1999). Urbanization in the Puget Sound region of Washington has also been associated with a decrease in plant-species richness (Azous and Richter 1995, Richter and Azous 1995, Reinelt et al. 1998, Reinelt and Taylor 2001).

Galatowitsch et al. (2000) observed that a reduction in native graminoid and herbaceous-perennial abundance at wet meadows in Minnesota coincided with recent cultivation, local urban-stormwater discharges, and the percentage of agriculture and urban land within a 500-m (1,640-ft) radius. The native vegetation was replaced with annuals in recently cultivated sites and by introduced perennials and duckweeds (lemnids) in stormwater-impacted meadows.

Jorgensen and Nauman (1994) found that plant-species composition in sedge meadows located adjacent to commercial-cranberry bogs in Wisconsin varied with distance from the bogs. The authors described many of the species found to decrease in importance (based on relative cover and frequency) with distance from the cranberry bogs as species that favor dry-sandy sites, a condition they attributed to sand blown from the bogs. Variations in plant-species composition in an adjacent *Sphagnum* bog were not as discernible.

Although not noted by Gibson et al. (1988) in their Pinelands forest-fragmentation study, the species found only in the forest fragments included trees not typically found in the central Pinelands, such as several species of hickory (*Carya* sp.), and edge or early successional species, such as sassafras (*Sassafras albidum*), red cedar (*Juniper virginiana*), gray birch (*Betula populifolia*), Virginia pine (*Pinus virginiana*), and black cherry (*Prunus serotina*), that are relatively common in well established residential and agricultural landscapes within the Pinelands. The authors do indicate that differences in the composition and structure between the fragmented and unfragmented stand types may be related to the proximity of residential development to the fragments and the higher incidence of disturbance in these stands, which they suggest are processes involved in forest fragmentation. In contrast, Guntenspergen and Levenson (1997) found no consistent relationship between the plant-species composition of upland-forest remnants and surrounding land use along an urban to rural gradient in Wisconsin. However, agriculture or old fields surrounded rural-forest stands, and all stands had been subjected to some type of human disturbance.

Urban land use can have impacts on oak forests beyond changes in woody-species composition associated with fragmentation. Baxter et al. (1999) found lower ectomycorrhizal richness on roots of mature red oak (*Quercus rubra*) in soil cores from urban sites compared to rural sites and attributed the differences to human-related impacts. Urban sites had higher nitrogen-deposition rates and heavy-metal levels in the soil.

Ehrenfeld (1983) was the first to compare the plant-species composition of Pinelands wetland forests located within developed and agricultural watersheds to that of undisturbed basins. Compared to undisturbed basins, a loss of characteristic-herbaceous species, establishment of nonnative species, and higher species richness characterized wetland forests in disturbed basins. The change in herbaceous species supports the premise that herbaceous-wetland plants may be better indicators of landscape change than slow-growing and long-lived woody plants because they may be respond more quickly to disturbance (Lopez et al. 2002), although dispersal and establishment also varies among herbaceous-plant guilds (Galatowitsch and van der Valk 1996).

Ehrenfeld (2005) also studied mature deciduous-wetland forests in urbanized northeastern New Jersey where she found that structural characteristics, common species, species richness, species composition, and the proportion of the flora represented by exotic species were similar to those reported for undisturbed red maple swamps in the glaciated region extending from northern New Jersey into central New Hampshire and Maine. She concluded that the evidence suggested forested wetlands in developed landscapes could maintain characteristics similar to those found in undeveloped landscapes. Although not stated by Ehrenfeld (2005), it is doubtful that this conclusion applies to Pinelands hardwood swamps, which share similarities to southern swamp communities (Ehrenfeld and Gulick 1981). Many of the herbaceous species that Ehrenfeld (2005) indicated are common in the northern New Jersey wetland sites are non-Pinelands species that are usually associated with the most heavily impacted Pinelands wetlands (Ehrenfeld 1983, Zampella and Laidig 1997, Zampella et al. 2001, 2003, 2005, 2006a).

The local effect of roads and adjacent development on the hydrology, water quality, and community composition and structure of Pinelands Atlantic white cedar swamps was examined by Ehrenfeld and Schneider (1990, 1991, and 1993). Changes in vegetation associated with suburbanization and road runoff included the loss of native species, the occurrence of non-Pinelands species, and a decrease in *Sphagnum* cover (typically associated with optimal cedar-seedbed conditions) and cedar germination. Laidig and Zampella (1999) evaluated the regional effect of upstream land-use disturbances on the same plant-community attributes in Pinelands Atlantic white swamps located within state-forest reserves in the Mullica River Basin. Cedar swamps that were located adjacent to streams in watersheds with greater than 40% altered land (developed and upland-agricultural land) generally had fewer understory plant species and differed in overall understory species composition from sites in watersheds with low (<10%) or moderate (10-20%) levels of altered-land cover. These differences were attributed to canopy conditions and regional differences in biogeography rather than the effects of watershed disturbance. Although the percentage of plants described by Stone (1911) as restricted to the Pinelands was significantly higher at sites in the least-altered watersheds compared to those in the most-highly altered watersheds, there was no significant difference between sites in the least- and the most-disturbed watersheds when the percentage of all native Pinelands plants, non-Pinelands plants, and facultative-upland species were compared. Virginia creeper (*Parthenocissus quinquefolia*) was the only non-Pinelands plant species associated with sites in the more highly altered watersheds, and plants that are not native to New Jersey were absent at all sites. *Sphagnum* cover was lowest at sites in the most altered watersheds, but there were no significant differences in overall seedbed conditions or cedar-seedling density between site types.

Laidig and Zampella (1999) attributed the differences in their results and those of Ehrenfeld and Schneider (1990, 1991, 1993) to the contrasting effects of local and regional land-use disturbances, and suggested that because Atlantic white cedar swamps depend primarily on groundwater, adjacent land uses rather than watershed-wide land-use patterns may have a greater influence on the plant-species composition of this wetland type. Unlike stream communities, cedar swamps within protected forest land located a distance from upstream disturbances and not affected by overbank flooding from adjacent streams appear to be buffered from the effect of land-use disturbances in a watershed.

A second Pinelands study also demonstrated the effect of adjacent land use versus watershed-wide land use on palustrine wetlands. Zampella and Laidig (2003) compared water-level patterns, water-quality, and vegetation-composition of natural ponds and small, excavated basins (borrow pits) located within state forests in the Mullica River Basin. Although several of the ponds were located in watersheds with high to moderate levels of development and upland agriculture and streams characterized by degraded water quality and the presence of non-Pinelands species (Zampella and Laidig 1997, Zampella and Bunnell 1998, Zampella et al. 2001), non-Pinelands plants were generally absent at all ponds. As with the Atlantic white cedar swamps studied by Laidig and Zampella (1999), the acid-water ponds situated within protected forest appeared to be buffered from distant land-use disturbances within the same watershed.

Community responses. In Minnesota riparian wetlands, shrub-carr vegetation (shrub-swamp vegetation), bird, and fish diversity and richness generally decreased with an increase in the extent of cultivated land in the surrounding landscape (Mensing et al. 1998). Each wetland community responded to land use at different scales. Shrub-carr richness was related to land use within 500 m (1,640 ft), and diversity was correlated with land use within both 500 m and 1,000 m (3,281 ft). Fish diversity and richness were correlated with land use within 2,500 m (8,202 ft) and at the watershed scale. Bird diversity was related to land use at both the 500-m and 1,000-m scales, with richness correlated with land use at the 100-m (328 ft) scale.

The Effect of Roads and Transmission-line Corridors on Ecological Integrity

As indicated by several of the previously cited studies, roads can have a substantial impact on ecological integrity. Road effects include mortality from road construction and vehicles, traffic noise, changes in animal behavior and movement, alteration of the physical and chemical environment, spread of exotic species, and increased use of areas by humans (Trombulak and Frissell 2000, Bennett 1991, Reijnen et al. 1995). Forest-interior species, species with low reproductive rates, and those with large-area requirements are most likely to be affected by the habitat loss that occurs with road construction (Forman et al. 2003).

Line corridors associated with human habitation, such as roads and roadsides, railroads, dikes, ditches, and power lines are dominated by edge species (Forman and Godron 1986). Malanson (1993) suggested that riparian zones are particularly accessible to wind-dispersed and animal-dispersed plant species where edge habitat is extensive and that wide riparian zones may present a barrier to the wind-dispersed seeds of upland plant species.

Species most affected by direct mortality on roads are those that are attracted to road habitat, are habitat generalists, and have high intrinsic mobility, multiple-resource needs, and large area requirements (Forman et al. 2003). Fahrig et al. (1995) found that the proportion of dead frogs and toads along Canadian roads increased with increasing traffic intensity. A majority of snakes observed on the main road in Everglades National Park by Bernardino and Dalrymple (1992) were injured or dead. Road mortality is suspected as being a contributing factor in the decline of some turtle species (Gibbs and Shriver 2002), with females killed disproportionately (Gibbs and Steen 2005, Steen et al. 2006). Carr and Fahrig (2001) found that the more-mobile leopard frog suffered reduced abundance due to increased traffic density while the less-mobile green frog was not affected. In a study of road mortality on a causeway adjacent to a wetland, Ashley and Robinson (1996) found that the road area adjacent to shallow ponds had the highest turtle mortality (Ashley and Robinson 1996).

Reinert and Zappalorti (1988) reported that gravid female timber rattlesnakes preferred the habitat conditions along sand-road edges in the Pinelands, which has a disproportionate effect on the population. Road mortality is frequently identified as a threat to two other Pinelands species, the red-headed woodpecker (*Melanerpes erythrocephalus*) and northern pine snake (*Pituophis m. melanoleucus*), although the actual effect of road kills has never been quantified.

Forman and Alexander (1998) indicated that although road kills may have local effects, they rarely limit population size and that road avoidance and barrier effects, which subdivide populations, may have greater ecological impacts. A translocation study of forest-dwelling bird species revealed that the movement of territorial, mated males in relation to barriers represented primarily by transportation corridors varied among a long-distance migrant, a short-distance migrant, and resident species, although return times were generally longer for individuals transported across the barriers (Bélisle and St. Clair 2001). In Manitoba, Canada, garter snakes usually avoided a 4-meter-wide gravel road with no vehicular traffic (Shine et al. 2004). Males were less able to follow pheromonal trails from females across the road than in the adjacent grassland. Andrews and Gibbons (2005) found that avoidance of roads in South Carolina varied by snake species. In that study, timber rattlesnakes either avoided crossing the road or were usually deterred after beginning to cross. When they did cross, they showed the slowest crossing speed of any of seven snake species tested. Timber rattlesnakes became immobile when a vehicle passed and frequently remained still for a minute or more afterwards.

Marsh and Beckman (2004) surveyed red-backed salamanders (*Plethodon cinereus*) and slimy salamanders (*P. glutinosus* and *P. cylindraceus*) at sites where gravel roads bisected mature forest in the Southern Appalachians of Virginia. Fewer red-backed salamanders were found near roads, which the investigators attributed to variations in soil moisture. Semlitsch et al. (2007) also attributed lower salamander abundance near low-use forest roads and abandoned-logging roads in a North Carolina hardwood forest with a decrease in the availability of moisture near the roads. They suggested that larger roads with a greater drying effect would result in a larger road-effect zone for salamanders. Gibbs (1998b) found that the movement of amphibians in the forest interior, across edges between forest and open land, and across forest-residential edges was greater than movement across forest-road edges.

Anderson et al. (1977) found that an increase in the width of transmission-line corridors dominated by mixed grass-annual-*Rubus* plant communities on bird species in Tennessee was associated with a decrease in forest species and an increase in open-country species. Rich et al. (1994), who conducted point-count bird surveys along road- and powerline-corridor transects in oak-pine and hardwood forests in Cumberland and Cape May counties in southern New Jersey, concluded that decreased abundances of the forest-interior birds at points along forest-dividing corridors that were 8 to 23-m (26 to 75-ft) wide were probably due to an overall decrease in forest habitat rather than avoidance of the narrow open-area corridors.

HYDROGEOLOGY

Groundwater Flow

Nearly all of the New Jersey Pinelands occurs on the Outer Coastal Plain and is closely associated with the Cohansey Sand. The Cohansey Sand, which is predominantly quartz sand and gravels with minor amounts of pebbly sands, finer sediments and interbedded clay, forms the upper part of the unconfined Kirkwood-Cohansey aquifer system (Rhodehamel 1979a, 1979b, Zapecza 1989). Local clay beds within the Cohansey Sand can be relatively thick. The Kirkwood Formation forms the lower part of the aquifer system (Rhodehamel 1979b, Zapecza 1989). Although the lithology of the Kirkwood Formation is variable, in the Pinelands the upper part is primarily sand and silty sand (Zapecza 1989). A discontinuous veneer of more-recent deposits, including the Beacon Hill Gravel, the Bridgeton Formation, and the Cape May Formation, overlies the Cohansey Sand and outcrop areas of the Kirkwood Formation.

The Kirkwood-Cohansey aquifer system is a water-table aquifer, although local deposits of interbedded clay may create locally confined or perched conditions. The aquifer exerts considerable influence on the Pinelands ecosystem, with the uplands and wetlands operating as a single hydrologic unit that is characterized by a largely unidirectional flow of water down elevational gradients (Ballard 1979).

Rhodehamel's (1979b) annual hydrologic budget for the Pinelands provides a simple model that relates precipitation to stream discharge. Total stream discharge represents 50% of precipitation. Groundwater discharge alone accounts for 89% of annual stream discharge in undeveloped watersheds. Direct runoff, which accounts for the remaining 11%, is episodic and associated primarily with wetlands. Although water yields of Pinelands streams are not uniform, Rhodehamel's (1979b) budget provides stream-discharge estimates that are generally comparable to those obtained by relating discharge and basin size of gauged streams (Figure 1.5) or from predictions based on correlating partial-discharge records with continuous-discharge data (Watt and Johnson 1992). In small headwater Pinelands streams, a higher percentage of infiltrating precipitation may follow a regional flow path which bypasses local streams and discharges to more distant streams (Rhodehamel 1979b).

Johnsson and Barringer (1993) estimated that approximately 8 to 13 cm (3 to 5 in) of groundwater recharge leaves the 6.09 km^2 (2.35 mi^2) McDonalds Branch basin in the regional-flow system. Interbasin transfer of water from adjacent watersheds can account for average annual flows that are higher or lower than predicted (Pinelands Commission 1980).

Rhodehamel (1979b) estimated near-surface groundwater velocity in the northwestern portion of Wharton State Forest to be 36.6 to 48.8 m yr^{-1} (120 to 160 ft yr^{-1}). Because recharge in upland areas follows deeper-flow patterns, groundwater-travel times from recharge areas in the Kirkwood-Cohansey increase with distance to stream courses. Discharge to streams and wetlands is also influenced by site-specific conditions. Johnsson and Barringer (1993) found varying groundwater/surface-water relationships along the length of McDonalds Branch. Water appeared to be draining to the shallow-groundwater system in sections of the stream channel and seasonal differences in recharge and discharge relationships were observed. Impervious materials beneath stream channels may also impede movement of water between the stream and groundwater (Lang and Rhodehamel 1963, Johnsson and Barringer 1993).

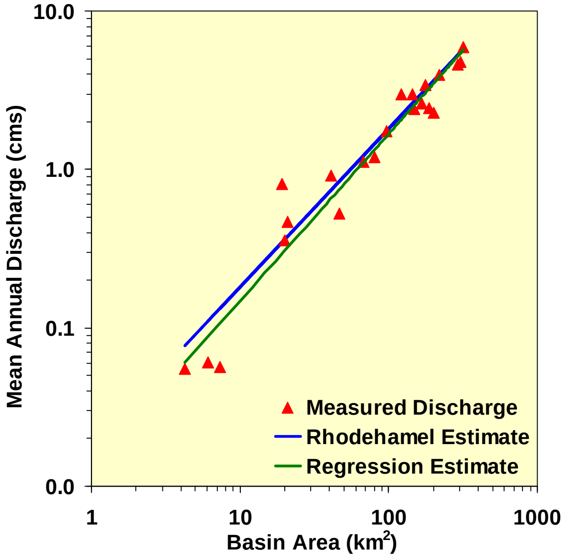

Figure 1.5. Relationships between watershed area and both measured and estimated mean annual-stream discharge. Mean annual-discharge values were based on U.S. Geological Survey records for 22 stream-gauging sites with periods of record ranging from 4-81 years.

The effect of variable groundwater-flow patterns and travel times on source areas of flow to streams is highlighted by several more recent Kirkwood-Cohansey studies. Modica (1996) used a groundwater-flow model to characterize groundwater-flow patterns and resident times in the Kirkwood-Cohansey aquifer in the upper parts of the Rancocas Creek and Wading River watersheds. He also simulated the effects of hypothetical-groundwater-withdrawal scenarios on streams. Groundwater residence time, which is defined as the period during which groundwater remains in the aquifer between the time it enters the system as recharge and the time it leaves as discharge, ranged from near zero to about 200 years. Most groundwater remained in the system for less than 20 years. Groundwater-residence times are related to aquifer thickness, with greater residence times in thicker parts of the aquifer near major basin divides where vertical-groundwater flow predominates. Conversely, horizontal flow predominates in thinner parts of the aquifer. Recharge in areas near major watershed divides was the source of flow to distant parts of the aquifer system, suggesting that source areas of groundwater flow to first-order streams do not coincide with topographic boundaries of watersheds. Groundwater-withdrawal simulations in the area of McDonalds Branch and Middle

Branch Mount Misery Brook indicated that withdrawals at the basin divide captured recharge that under natural conditions would flow deep into the aquifer and had little effect on adjacent streams, whereas withdrawals away from the divide captured more recharge from source areas of flow to the streams. Similarly, groundwater modeling of the northeastern part of the Mullica River Basin revealed that some recharge captured near the headwaters discharged to stream reaches located relatively far down the watershed (Modica 1998).

In contrast to the results obtained in the Rancocas River and Mullica River basins, Modica et al. (1998) indicated that the source area of groundwater flow to the upper reach of the Cohansey River and its tributaries extended to the basin divide throughout the upper part of the watershed. They suggested that the stream probably captured all the recharge entering the upper part of the watershed because the Kirkwood-Cohansey aquifer in that area is relatively thin. The source area of Barretts Run, a Cohansey River tributary located in the lower part of the watershed, did not extend to the drainage divide. Some groundwater originating near the drainage divide flowed under Barretts Run and discharged to the Cohansey River. Additionally, most of the source area of flow to Barretts Run was from one side of the stream, which was attributed to the fact that Barretts Run flows at an angle to the direction of groundwater flow. The source area of flow to Clarks Run, a headwater tributary of the Cohansey River that flows parallel to the direction of groundwater flow, was almost evenly distributed on both sides of the stream.

Simulation of groundwater flow in a hypothetical unconfined aquifer with properties that are typical of the Atlantic Coastal Plain, including the Coastal Plain of New Jersey, demonstrated that as aquifer thickness decreased, the source area of flow to a hypothetical stream increased exponentially and start-of-flow occurred closer to the upgradient divide (Modica et al. 1997). Because the depth to which groundwater can descend in thin aquifers is limited, the extent of source areas becomes relatively large. Modica et al. (1997) concluded that discharge to a point in a stream originates from sources that are near and far from the stream, with flows that originate close to the edge of the source area following longer and deeper paths than flows that originate closer to the stream. Using a groundwater-flow model, particle-tracking analysis, and groundwater samples collected from stream transects in the Cohansey River Basin, Modica et al. (1998) corroborated a conceptual model that also indicated that groundwater discharging from the Kirkwood-Cohansey aquifer to a stream originates from sources that are near and far from the stream. Based on the results of several Kirkwood-Cohansey groundwater studies, Szabo et al. (2005) indicated that most water recharged near wetlands is discharged to the wetlands within 5 to 10 years.

Stream Flow

Two studies assessed the potential impact of groundwater withdrawals on Pinelands streams. Cauller and Carleton (2006) used a groundwater-flow model to simulate the effect of groundwater withdrawals from the unconfined Kirkwood-Cohansey aquifer system on stream flows in the upper part of the Maurice River Basin. A comparison of predevelopment conditions with post-development (1995-97) conditions revealed a significant reduction in baseflows in Scotland Run associated with withdrawals from nearby public-supply wells. Simulations using hypothetical public-supply wells produced a nearly one-to-one correlation between the amount of groundwater withdrawn and the amount of baseflow reduction in Scotland Run when wells were located adjacent to the stream. Placing the hypothetical wells on a topographic divide distributed the baseflow reduction between Scotland Run and the adjacent stream. Nicholson and Watt (1997) evaluated the effect of groundwater withdrawals from Kirkwood-Cohansey wells on baseflows in the Toms River, Metedeconk River, and Kettle Creek. Simulation results indicated that groundwater withdrawals during the 1980’s reduced predevelopment baseflow in some streams by as much as 11% and caused average water-level declines up to about 20 ft (6.1 m).

Dow (2007) related relative-stream-baseflow volume to land use in nine Pinelands watersheds. Four of the more urbanized watersheds, with data records of 72 to 74 years in length, were found to have significant, yet small changes in stream flow, with the Toms River (1929-2001) and Great Egg Harbor River (1927-2001) showing a decrease in relative-baseflow volume and the Batsto River (1928-2001) and North Branch Rancocas Creek (1927-2001) showing the opposite trend. Relative-baseflow volume decreased by about 3% in the Toms River and by about 2% in the Great Egg Harbor River. Baseflow increased by about 3% in both the Batsto River and North Branch Rancocas Creek. Fluctuations in stream flow relative to total stream flow over individual year-long periods were higher for the Toms River and Great Egg Harbor River (i.e., these streams were flashier). Dow (2007) attributed the difference in trends to a slow down in urbanization and a change in wetland agricultural practices in the Batsto River and North Branch Rancocas Creek watersheds. However, he did indicate that interpreting significant trends in the face of variable and sometimes undefined landscape changes is a challenge. Regarding wetland agriculture, baseflow decreased as the percentage of wetland agriculture in a watershed increased and flashiness increased as the percentage of artificial lakes/reservoirs increased.

WATER QUALITY

Water-quality Patterns in the United States

Agriculture and urban activities are a major source of nitrogen and phosphorus to surface and groundwaters of the United States (Carpenter et al. 1998, Nolan 2001). In agricultural areas, water-quality degradation is associated with the application of lime and fertilizers (Hamilton and Shedlock 1992, Hamilton et al. 1993). In urban areas, where sewage discharges are a major source of point-source pollution (Osborne and Wiley 1988, Sliva and Williams 2001), nonpoint source pollution is generally related to the amount of impervious surface in a watershed (Klein 1979, Arnold and Gibbons 1996).

Groundwater quality. Anderson (1993) found that concentrations of nitrite plus nitrate nitrogen in the Anoka Sand Plain Aquifer of Minnesota were related to agricultural and residential land uses, with median concentrations of 0.22, 2.0, 5.3, and 4.2 mg L^{-1} for undeveloped, unirrigated-cultivated, irrigated-cultivated, and residential lands, respectively. Hamilton et al. (1993) reported that concentrations of nitrate nitrogen in water samples collected from the surficial aquifer in agricultural areas throughout the Delmarva Peninsula ranged from 0.4 to 48 mg L^{-1}, with a median concentration of 8.2 mg L^{-1}. The 0.4 mg L^{-1} was an estimated-threshold value established to represent natural groundwater in the area. Eckhardt and Stackelberg (1995) compared water-quality in the unconfined upper-glacial aquifer of Long Island, New York in relation to sewered and unsewered suburban, agricultural, and forested land. Nitrate concentrations in samples from the suburban and agricultural areas were higher than those found in samples from the undeveloped-forested area.

Ritter and Chirnside (1984) reported that nitrate concentrations in the water-table aquifer of southern Delaware were highest in areas with intensive-poultry production and areas with intensive-crop production and excessively drained soils. In central Pennsylvania, Pionke and Urban (1985) observed that nitrates, chloride, and phosphate concentrations were much higher in groundwater underlying cropland than concentrations in groundwater underlying forests.

Surface-water quality. Degradation of streams draining urban land has recently been described as the “urban stream syndrome” (Meyer et al. 2005). Symptoms of the syndrome include increased stream flashiness, water-quality degradation, altered-channel morphology, a reduction in species richness, and an increase in tolerant species (Walsh et al. 2005).

The impact of nutrient enrichment on stream ecosystems is especially significant (Smith et al. 1999). Jordan et al. (1997a, 1997b) related nutrient concentrations in Piedmont and Coastal Plain streams throughout the Chesapeake Bay drainage to the percentage of agricultural land in the associated watersheds. Nitrate concentrations increased as the percentage of agriculture in a watershed increased. Phosphorus concentrations, which did not vary in relation to land use, correlated with particulate matter concentrations, indicating that discharge of phosphorus is related to sediment transport. Jordan et al. (1997c) suggested that groundwater was the major source of nitrate in streams. Although pH and alkalinity varied between Coastal Plain regions, neither correlated with land use (Jordan et al. 1997a). During storm events, total nitrogen, ammonium, nitrate, particulate organic phosphorus, and particulate phosphate concentrations discharging to Chesapeake Bay were highest in a Coastal Plain watershed dominated by row crops compared to three other watersheds characterized as mixed land use, primarily grazed pasture, and primarily forest (Correll et al. 1999). Liu et al. (2000) found that an increase in nitrate-nitrogen concentrations was associated with the extent of cropland in streams across all physiographic provinces of the Chesapeake Bay drainage.

Tufford et al. (2003) evaluated the effects of urbanization on nutrients in several small Coastal Plain streams in South Carolina. Nitrate and total phosphorus concentrations were higher in urban streams compared with forested streams. In contrast, dissolved-organic nitrogen and ammonium were higher in the forested streams. The high dissolved-organic nitrogen in the forested streams was attributed to the naturally high organic content associated with blackwater streams. Although nutrient concentrations differed between stream types, multiple regression models revealed no significant relationship between land use and nutrients, which was attributed to the small size of the watersheds studied.

In Iowa, Schilling and Libra (2000) found that nitrate-nitrogen concentrations in surface waters was directly related to the percentage of row-crop land in a watershed. They concluded that watershed size influenced this relationship, with smaller watersheds displaying a greater increase in nitrate-nitrogen as the percentage of row-crop land increased. Streams in an east-central Pennsylvania watershed draining a forested ridge exhibited low ion concentrations, including nitrates, compared to streams originating in agricultural areas that included crop land, pasture, and confined-animal operations (Gburek and Folmar 1999). Based on a comparison of baseflow and groundwater chemistry, Gburek and Folmar (1999) suggested that land use controlled both baseflow and groundwater quality at the subwatershed scale in their study area. In mid-Atlantic streams where agriculture and forest were the dominant land uses, Herlihy et al. (1998) reported that elevated nitrate concentrations are closely associated with agriculture and that chloride appears to be a general indicator of any non-forest land.

Rhodes et al. (2001) found a positive relationship between stream-water nitrate and sulfate concentrations and the percentage of developed and agricultural land in a Massachusetts watershed. They also found that chloride concentrations increased with road density. Johnson et al. (1997) associated the highest nitrite plus nitrate concentrations, total dissolved solids, and alkalinity in Michigan streams with watersheds dominated by row-crop agriculture. Urban land was also associated with increases in total dissolved and suspended solids. Factors other than land use appeared to regulate phosphorus concentrations in the streams. In the Etowah River basin of Georgia, both agricultural- and urban-land cover were positively associated with turbidity, ammonium-nitrogen, and nitrite plus nitrate-nitrogen in streams (Roy et al. 2003). Urban-land cover was also positively correlated with soluble reactive phosphorus, specific conductance, and total suspended solids

Turfgrass management can also impact surface waters. King et al. (2001) measured nitrite plus nitrate-nitrogen, ammonium-nitrogen, and orthophosphate in a stream entering and exiting a golf course in Texas. Median baseflow and storm-flow nitrite plus nitrate-nitrogen concentrations exiting the golf course were greater than concentrations entering the area. Median baseflow and storm-flow concentrations increased by 0.30 mg L^{-1} and 0.86 mg L^{-1}, respectively. Orthophosphate concentrations remained relatively constant and ammonium concentrations decreased as the stream passed through the course.

Water-quality Patterns in the Pinelands

Groundwater quality. Nitrate movement into groundwater is a special concern in the southeastern Coastal Plain, which includes the Pinelands (Hubbard and Sheridan 1989). Since the 1940's, nitrogen loading, based on sales of nitrogen fertilizer, has increased dramatically in parts of southern New Jersey (Kauffman et al. 2001). Analysis of nitrogen data collected from shallow wells in the Kirkwood-Cohansey aquifer indicated that concentrations of nitrate nitrogen were higher in agricultural areas than undeveloped areas and concentrations increased as the percentage of agricultural land in the vicinity of a well increased (Vowinkel and Tapper 1995). Stackelberg et al. (1997), Szabo et al. (1997), and Watt and Johnson (1992) reported elevated nitrate concentrations in water samples taken from Kirkwood-Cohansey wells in agricultural areas. Stackelberg et al. (1997) found that median nitrate nitrogen concentrations in the Kirkwood-Cohansey aquifer varied in relation to the type of land-use. The median concentrations in undeveloped areas was 0.07 mg L^{-1} compared to median concentrations of 2.6 mg L^{-1} in recently developed areas, 3.5 mg L^{-1} in older developed areas, and 13 mg L^{-1} in agricultural areas.

Based on modeling and water-quality monitoring, Modica et al. (1998) demonstrated that contaminants present in the Kirkwood-Cohansey aquifer reflected the land-use activities associated with the time of origin of the groundwater. Both modeling and water-quality monitoring results indicated that most groundwater discharging to the Cohansey River was young enough to be affected by nitrogen from surrounding agricultural lands. Groundwater age, which represents the time since recharge entered the aquifer, was based on the concentration of chlorofluorocarbons (Freon compounds). Chlorofluorocarbons can be used to determine groundwater age by relating groundwater concentrations to known historical atmospheric concentrations. Nitrate concentrations were highest in water recharged since about 1986. Groundwater discharging near the center of the stream channel was older and with lower nitrogen concentrations compared to water discharging near the stream bank.

A subsequent Kirkwood-Cohansey hydrologic-modeling study also indicated that nitrate concentrations in streams and wells were related to land use and groundwater travel time. Kauffman et al. (2001) found that nitrate concentrations were lowest in undeveloped areas and highest in agricultural areas. The time needed for water to travel from groundwater recharge to stream-discharge points varied with distance to a stream and ranged from greater than 200 years near watershed divides to less than 50 years near streams.

Surface-water quality. Water-quality degradation in Pinelands streams is directly related to basin-wide upland land uses and is due primarily to nonpoint source pollution (Dow and Zampella 2000, Hunchak-Kariouk and Nicholson 2001, Morgan and Good 1988, Zampella 1994, Zampella et al. 2001, 2007a, Conway 2007). Specific conductance, pH, and concentrations of calcium, magnesium, chloride, sulfate, and nitrite plus nitrate-nitrogen increase as the percentage of developed (urban) and upland-agricultural lands in a watershed increases. In contrast, streams draining forested Pinelands watersheds are usually very acidic with low concentrations of dissolved solids.

In the Pinelands, impervious surface, which is associated with nonpoint source pollution, is proportional to the extent of developed land in a watershed (Conway 2007, Figure 1.6). However, to accurately account for variations in water chemistry in Pinelands streams, the effect of both developed lands and upland agriculture must be considered (Conway 2007, Zampella et al. 2007a). Morgan and Good (1988) reported elevated pH and calcium, magnesium, potassium, and sulfate concentrations in Mullica River Basin streams draining watersheds disturbed by development and agriculture compared to undisturbed streams. Although the effect of land-use activities on water quality is most obvious when comparing reference streams to streams in highly altered watersheds, variations in water quality represent a continuum related to a gradient of increasing watershed disturbance associated with developed land and upland agriculture (Zampella 1994, Zampella et al. 2007a).

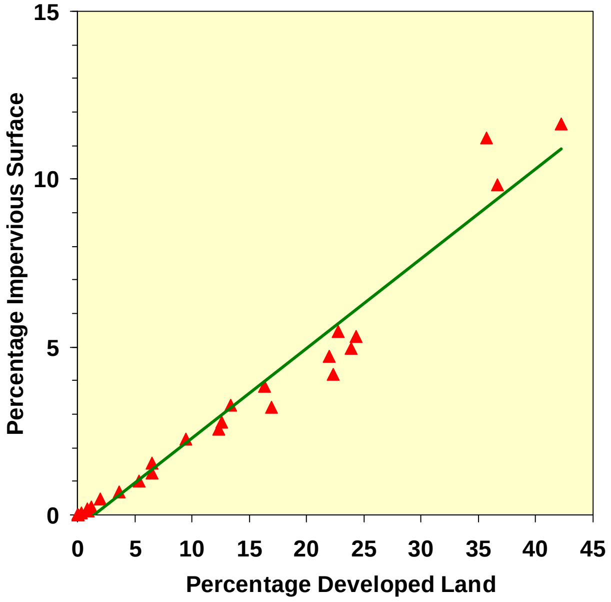

Figure 1.6. The relationship between developed land and impervious surface in the Mullica River Basin. The relationship is given for watersheds associated with the 25 U. S. Geological Survey water-quality monitoring sites studied by Zampella et al. (2007a). Source of land use data: NJDEP (2000).

Zampella et al. (2007a) used multiple regression and water-quality data for 25 stream sites in the Mullica River watershed, where nonpoint sources are the main contributors of pollutants to surface waters, to determine the percentage of variation in pH, specific conductance, calcium, magnesium, chloride, sulfate, nitrate, and phosphorus explained by developed land and upland agriculture. A second, independently collected water-quality-data set was used to validate the statistical models. The analyses revealed a statistically significant relationship between water-quality degradation and basin-wide upland land uses and demonstrated that the percentage of both developed land and upland agriculture are generally good predictors of water-quality conditions. Wetland agriculture, which includes blueberry and cranberry farms, was not a significant predictor of water quality. The threshold at which a significant deviation from reference-site water-quality conditions consistently occurred was <10% altered-land (developed land and upland agriculture) cover in a basin.

Dow and Zampella (2000) related specific conductance and pH measured at 45 Pinelands stream sites to the percentage of altered land (developed land and upland agriculture) in a watershed. Both variables were positively associated with increases in the extent of altered land. Altered land explained 56% of the variability in specific conductance and 48% of the variability in pH. The water-quality/land-use relationships did not vary significantly between major Pinelands watersheds.

Conway (2007) also found that land-use patterns in Pinelands watersheds are associated with variations in pH and specific conductance in Pinelands streams.

Similar water-quality/land-use patterns have been observed in watershed studies conducted in the four major Pinelands watersheds, including the Mullica River, Rancocas Creek, Great Egg Harbor River, and Barnegat Bay watersheds (Table 1.3). Although the relationship between pH and altered land was weakest in the Barnegat Bay watershed, the water-quality/land-use relationships for individual stream systems in the Barnegat Bay study area were stronger than those revealed when all stream-monitoring sites in the area were included in a single analysis and were similar to the relationships found in the other three major watersheds studied (Zampella 2006a). The strength of the relationship between specific conductance and land use was similar in the Barnegat Bay, Mullica River, and Great Egg Harbor River basins and strongest in the Rancocas Creek basin (Table 1.3).

Table 1.3. Spearman rank correlations (r) relating pH, specific conductance (SC, μS cm^{-1}), and nitrite plus nitrate as nitrogen (NO_x) to the percentage of altered land (developed land and upland agriculture) in a stream basin for four major Pinelands watersheds. The p value is < 0.001 for all correlations except Great Egg Harbor River NO_x, where $p = 0.003$. The sample size (n) represents the number of stream-monitoring sites included in each analysis. From Zampella et al. (2006a).

		pH	SC		NO_x
Watershed	n	r	r	n	r
Mullica River	103	0.83	0.59	25	0.77
Rancocas Creek	49	0.86	0.84	51	0.50
Great Egg Harbor River	49	0.79	0.55	12	0.77
Barnegat Bay	58	0.60	0.57	20	0.78

Elevated nitrite plus nitrate nitrogen concentrations in Pinelands streams have been associated with the extent of developed land and upland agriculture throughout the region (Zampella 1994, Zampella et al. 2006a, Zampella et al. 2007a, Table 1.3). In the absence of point-source discharges, concentrations of both ammonia and phosphorus are usually low or below detection, even in streams draining areas with relatively extensive areas of developed land and upland agriculture (Morgan and Good 1988, Zampella 1994, Zampella et al. 2007a). Elevated ammonia concentrations have been associated with altered land only in the Rancocas Creek, and no clear relationship between altered land and phosphorus concentrations has been demonstrated in any of the four major watersheds (Zampella 1994, Zampella et al. 2006a, Zampella et al. 2007a).

Sources of water-quality degradation. Geology can have a significant effect on water-quality (Patrick 1996, Liu et al. 2000), but land use can overshadow the effect of geology where urban and agricultural areas are extensive (Leland and Porter 2000). Unlike most freshwater systems, weathering of sediments is not a major source of dissolved solids in Pinelands streams (Yuretich et al. 1981, Morgan and Good 1988). In the Pinelands, where all watersheds share a similar underlying geology characterized by unconsolidated sands and gravels with variable amounts of silt and clay (Newell et al. 2000, Rhodehamel 1979a, Zapecza 1989), developed land and upland agriculture are the primary factors influencing water quality.

The higher pH in degraded Pinelands waters may be related to higher primary productivity and nitrate assimilation in eutrophic waters (Morgan 1985) or, as reported from other regions of the United States (Herlihy et al. 1998), to increases in calcium and magnesium. Elevated specific conductance associated with land use reflects an increase in the concentration of dissolved solids, including calcium, magnesium, chloride, and sulfate (Morgan and Good 1988, Zampella 1994, Zampella et al. 2007a). Yuretich et al. (1981) identified deep groundwater contributions as a possible source of elevated calcium and magnesium levels in both the Mullica River and the Batsto River. Morgan and Good (1988), who more accurately characterized the extent of developed land and upland agriculture in these watersheds, attributed the elevated concentrations of both cations to watershed disturbance. Liming is a potential source of both calcium and magnesium, which are found in elevated concentrations in the Kirkwood-Cohansey aquifer beneath agricultural lands (Watt and Johnson 1992, Johnson and Watt 1996, Szabo et al. 2005). Hamilton et al. (1993) also related liming of soils with higher concentrations of calcium and magnesium in groundwater associated with agricultural areas in the Delmarva Peninsula.

Marine aerosols are a source of sodium and chloride in both disturbed and undisturbed Pinelands streams (Morgan and Good 1988, Yuretich et al. 1981), but observed increases in chloride associated with increasing watershed disturbance probably reflects land-use patterns rather than atmospheric deposition (Zampella et al. 2001, 2007a). Chloride concentrations in streams are related to land use throughout the mid-Atlantic region (Herlihy et al. 1998). Liu et al. (2000) found that both sodium and chloride concentrations in Piedmont and Coastal Plain streams of the Chesapeake Bay drainage increased with the extent of urban land. Potential sources of chloride include the use of road salts in developed areas (Hay and Campbell 1990, Robinson et al. 1996, Liu et al. 2000, Rhodes et al. 2001, Kaushal et al. 2005) and septic systems (Bunnell et al. 1999, Liu et al. 2000). Atmospheric deposition is also a source of sulfate in the Pinelands (Morgan 1991), however as with sodium and chloride, the effect of land use appears to overshadow the atmospheric deposition because sulfate concentrations in stream water increases with the extent of developed and agricultural lands in a watershed (Zampella et al. 2007a).

Vowinkel and Tapper (1995) concluded that chemical fertilizers, rather than livestock waste or septic systems, were the predominant source of nitrogen in Kirkwood-Cohansey wells in an agricultural area. Jacobsen (2000) demonstrated the potential impact of fertilizers on groundwater in the Pinelands when he detected elevated nitrates in groundwater beneath an abandoned sand mine in Ocean county following a single experimental application of nitrogen at a relatively low rate (50 pounds per acre = 56.1 kg per ha). Nitrogen was applied as either commercial fertilizer or composted sludge (biosolids).

Agricultural fertilizers have a secondary impact on groundwater in the Kirkwood-Cohansey aquifer system. Several studies have indicated that chemical processes associated with the agricultural application of nitrogen, calcium, and magnesium are responsible for the mobilization of radium from sediments of the Bridgeton Formation (Kozinski et al. 1995, Szabo et al. 1997, 2005). Hydrogen ions present as a result of nitrification (transformation of ammonia to nitrite and nitrate) and divalent calcium and magnesium ions from soil additives compete with radium for sorption sites on the sediments (Szabo 2005).

Septic systems are another source of nitrates in the Pinelands (Bunnell et al. 1999) and other parts of the United States (Gold et al. 1990, Robertson et al. 1991, Harman et al. 1996, Steffy and Kilham 2004). Although there is evidence that some nitrate is adsorbed to unsaturated-zone sediments derived from the Bridgeton Formation (Reilly and Baehr 2006), this nutrient is fairly conservative and mobile (Robertson et al. 1991, Harman et al. 1996, Kauffman et al. 2001). Bunnell et al. (1999) found no significant decrease in nitrogen, consisting primarily of nitrate, between the top and bottom zones of the disposal bed of most Pinelands septic systems that they studied. Robertson et al. (1991) studied a 12-year old and 1.5-year old septic system serving separate single-family homes on a shallow unconfined-sand aquifer in Ontario. Nitrate concentrations at the end of a 130-m (427-ft) plume extending from the 12-year-old system were estimated to be 50% of the source concentration. The plume from the second system discharged to a river located 20 m (66 ft) away. Nearly complete nitrogen attenuation, which was attributed to denitrification, occurred almost immediately before discharge to the river. Based on their field results and modeling, Robertson et al. (1991) estimated that approximately 2 km (1.2 mi) would be required to reduce the source nitrate-nitrogen concentration to 2.5 mg L^{-1}.

Although groundwater is likely a major source of nitrates in Pinelands streams, stormwater runoff may also be an important source in more heavily developed areas. In their study of four Toms River tributaries, including Long Swamp Creek, Wrangle Brook, Davenport Branch, and Jakes Branch, Baker and Hunchak-Kariouk (2006) found that although mean concentrations of nitrate in baseflow and storm-flow samples were similar, nitrate concentrations increased during the rising limb and peak flow at the more heavily urbanized Long Swamp Creek, where developed land covered 72.4% of the watershed. Ammonia concentrations in Long Swamp Creek were also highest during the rising limb. Elevated ammonia levels in Pinelands streams have also been associated with direct sewage discharges (Fusillo 1981, Schornick and Ram 1998, Zampella 1994).

Nitrogen-removal Mechanisms

Plant uptake and denitrification are the two most likely mechanisms responsible for reducing nitrate contamination of groundwater (Peterjohn and Correll 1984, Correll 1996, Kauffman et al. 2001), but the role of both mechanisms, especially that of vegetation, is somewhat uncertain (Hill 1996). Peterjohn and Correll (1984) determined that cropland in Maryland’s Rhode River watershed was a major source of nitrogen and phosphorus to the riparian forest. Most nitrogen loss was in subsurface groundwater flow and total phosphorus loss was evenly distributed between surface and subsurface losses. The estimated nitrogen retention by riparian forest was 89%. Correll et al. (1992) estimated that agriculture in this Coastal Plain watershed accounted for 69% of the total nongaseous-nitrogen loads and 93% of the total phosphorus loads to the watershed. Cropland discharged more nitrogen per hectare in runoff compared to upland forest and pasture. Riparian-hardwood forest bordering the cropland removed over 80% of the nitrate and total phosphorus in overland flows and about 85% of the nitrate in shallow groundwater. Lowrance et al. (1984) and Snyder et al. (1998) also found that riparian-forest-buffer zones reduced groundwater-nitrate concentrations originating from upland-agricultural fields and discharging to an adjacent stream, and Cooper et al. (1987) and Correll (1996) indicated that riparian vegetation might play an important role in trapping particulates from overland sheet flow, especially in agricultural areas.

Riparian zones may be effective in removing nitrate from groundwater in areas with shallow, lateral-groundwater-flow paths, but they may be less effective where water flows mainly across the surface or in sand at depths where little interaction with sediments and vegetation occurs (Hill 1996). Correll (1996) also concluded that the effectiveness of riparian zones in improving water quality depends on the volume and pathway of water movement through the zone. Water passing beneath the riparian zone cannot interact with the sediments and vegetation.

In urban areas where surface runoff may dominate, the denitrification potential of surface soils can be high (Groffman and Crawford 2003). Hanson et al. (1994) estimated that denitrification removed approximately 59% of groundwater nitrate that entered a riparian-forest site located below and intensive-residential development with on-site septic systems in Rhode Island. However, unsaturated conditions associated with lower water tables in urban and suburban riparian zones may result in an increase in nitrification and a decrease in denitrification (Groffman et al. 2002).

Using multivariate logistic regression in a national-scale analysis, Nolan (2001) identified several variables that influence contamination of shallow groundwater by nitrate. The likelihood of contamination was positively associated with the combined effect of nitrogen-fertilizer loading, the percentage of cropland-pasture, population density, the percentage of well-drained soils, depth to the seasonally high water table, and the presence or absence of a fracture zone within an aquifer associated with the sampling wells. Nolan (2001) suggested two possible explanations for the positive relationship between an increase in the likelihood of nitrate contamination and depth to the seasonally high water table. First, a very shallow depth to groundwater creates the anoxic conditions needed to promote denitrification, thus reducing nitrate contamination of groundwater. Second, agricultural land, which is a source of nitrates, is more likely to be found on areas with greater depth to groundwater. However, the Spearman rank correlation between percentage cropland-pasture and depth to the seasonally high water table given by Nolan (2001) was quite low (r = 0.19), indicating that the relationship was weak. Bachman (1984) also suggested that denitrification may be responsible for lower groundwater-nitrate concentrations at sites in the Delmarva Peninsula with poorly drained soils. The possible role of denitrification is supported by Trudell et al. (1986), who obtained experimental evidence indicating that denitrification occurred in a shallow, unconfined aquifer in southwestern Ontario, Canada and by Smith and Duff (1988) who demonstrated that denitrification occurred in a sand and gravel aquifer. Although dissolved oxygen concentrations in Kirkwood-Cohansey groundwater are generally too high for denitrification to occur, and groundwater may follow deep paths through sands, Kauffman et al. (2001) estimated that in-stream processes or denitrification near streams removed about 40 percent of the nitrate in the Kirkwood-Cohansey aquifer near Glassboro, New Jersey.

Brown and Thomas (1978) conducted a small-scale experimental study that evaluated the uptake of nitrogen from septic-tank effluent by common bermudagrass (*Cynodon dactylon*). They placed sandy loam, sandy clay, and clay soils in separate lysimeters, planted bermudagrass, and pumped septic-tank effluent below the surface of the soils. Bermudagrass growing immediately above and adjacent to the septic line was periodically harvested and the average nitrogen uptake by the grass was determined. Estimates of nitrogen taken up by bermudagrass on the sandy loam, sandy clay, and clay soils were equivalent to 8.9, 32.2, and 45.6% of the applied amounts. Brown and Thomas (1978) concluded that cutting and removing grass from large septic fields with the less-permeable soils could remove a large percentage of nitrogen.

Based on the results of the experimental-lysimeter study, Brown (1980) included vegetal uptake in a Pinelands dilution model currently used to determine the land area required to dilute nitrogen from septic-tank effluent to an acceptable concentration. Estimating that the growing season in that study was about twice as long as that in New Jersey, Brown (1980) assigned a vegetal uptake value of 4.5% to soils in hydrologic group A (well-drained to excessively drained sands and/or gravels with high infiltration rates such as Lakewood soils). Although a reason was not given, a vegetal-uptake value of 9% was assigned to Group B soils (moderately well to well-drained soils with moderately fine to moderately coarse textures such as Downer soils). Brown (1980) indicated that vegetal uptake has a small effect on dilution area and that harvesting of the vegetation would be needed to remove nitrogen from the area.

Ehrenfeld (1987) studied the ability of woody vegetation to remove nitrogen from septic-tank leachate in the Pinelands. She concluded that although native-woody vegetation is capable of removing nitrogen from effluent, uptake and storage is low and vegetation at naturally occurring densities cannot significantly affect water quality. Ehrenfeld (1987) noted that the results of her study do not support the inclusion of vegetal uptake of nitrogen in models used to determine housing densities in the Pinelands.

Effect of the Proximity of Land Use on Surface-water Quality

The effect of the proximity of a land use on water quality has been addressed in several water-quality studies. In general, accounting for proximity yields results similar to those obtained using the simple proportion of developed or agricultural land in a watershed. Omernik et al. (1981)

evaluated whether the proximity of agriculture and forested land to streams influenced nitrogen and phosphorus levels at a nationwide network of stream-monitoring sites not affected by point-source pollution. Their results indicated that incorporating the proximity of agriculture and forested land to streams did not improve the predictive ability beyond that attributed to the proportion of these land uses in the total watershed.

In a study of Michigan streams, Johnson et al. (1997) found that the variability in stream chemistry explained by land use within 100-m (328-ft) stream buffers and in the entire watershed were similar. They attributed the similarity to the highly modified nature of the watersheds, noting that near-stream areas may simply reflect the dominant land use in the watershed. Johnson et al. (1997) concluded that relatively coarse spatial data could be used to describe regional water quality.

Hunsaker et al. (1992), who used multiple regression to relate conductance in streams to land use in entire watersheds and within 200 m and 400 m (656 ft and 1,312 ft) of stream corridors, found that the near-stream models did not improve the relationship between land use and conductance beyond that described by the watershed-level model. Hunsaker and Levine (1995) concluded that total nitrogen and total phosphorus concentrations in Illinois and Texas watersheds were related to the proportion of different land uses. In the Illinois study, proximity to streams was not an important factor, whereas the location of various types of land use was an important consideration in modeling water-quality impacts in Texas.

Osborne and Wiley (1988) reported that the effect of agriculture on soluble reactive phosphorus in a major Illinois watershed was minimal compared to streams in urban areas with point (sewage) and nonpoint sources. They found little difference in the variability in nitrate-nitrogen and soluble reactive phosphorus explained by land use within distances ranging from <100 to >1,000 m (<328 to >3,281 ft) from streams. Sliva and Williams (2001) evaluated the relationship of water quality in Ontario streams to both whole-watershed land use and land use within 100 m. Urban land rather than agriculture appeared to have a greater effect on concentrations of ammonium, fecal coliform, chloride, and total solid concentrations, and watershed-level land use had a slightly greater influence on water quality than land use located within 100 m. As with the Osborne and Wiley (1988) study, the effect of urban land was associated with both point (sewage) and nonpoint sources.

In the Maryland Coastal Plain streams studied by King et al. (2005), the percentage of cropland in a watershed was a strong predictor of nitrate-nitrogen. The use of inverse-distance-weighted cropland values did not improve predictions of nitrate-nitrogen concentrations over the raw basin-wide percentage of cropland values across all watersheds, in large watersheds (>2600 ha), or in medium watersheds (600-2600 ha), but was a better predictor of nitrate-nitrogen concentrations in small watersheds (<600 ha).

Proximity was found to be important in two other studies. Basnyat et al. (1999) reported that nitrate-nitrogen concentrations in Alabama streams were related to near-stream land uses rather than land uses in the whole watershed. The regression models developed by Basnyat et al. (1999) identified urban lands as a strong contributor of nitrate-nitrogen, with active agriculture as the second-largest contributor. In a study of eight, low-order watersheds with dominant land-uses that included agriculture, forest, urban land, and wetlands on the Coastal Plain of South Carolina, Tufford et al. (1998) reported that land uses close to the stream channel were a better predictor of total nitrogen and total phosphorus than land uses farther away from the channel.

Zampella et al. (2007a) evaluated the effect of the proximity of land use on Pinelands stream-water quality using a distance-weighted approach and multiple regression. Including the proximity of developed and agricultural land to a monitoring site using inverse-distance-weighted land-use values and a digital-elevation model to estimate flow paths did not improve the relationship between land use and water quality derived using basin-wide land-use data. One reason is that both the watershed-wide urban-land and upland-agriculture values were correlated with their respective distance-weighted values. A separate analysis using the same water-quality and land-use data analyzed by Zampella et al. (2007a) is summarized in Table 1.4. Spearman rank correlation was used to relate water quality to watershed-wide raw and distance-weighted altered land (developed land and upland agriculture), watershed-wide impervious-surface cover, and altered land within cumulative buffers (i.e., 0-100, 0-200, etc.) encompassing each stream. As shown in Table 1.4, the relationships that account for the proximity of altered land are similar to those obtained using watershed-wide land-use data. Furthermore, with the exception of chloride, variations in water quality are more closely associated with land-use compared to impervious-surface cover, which does not include upland-agricultural land. The similarity in the land-use/water-quality relationships obtained using different scales is associated with significant correlations among the scales, with Spearman rank correlation coefficients (r) ranging from 0.94 to 1.00. Although the percentage of altered land is generally lower in buffers near the streams and increases with increasing buffer width, the rank-order relationship between streams remains fairly constant regardless of buffer width (Figure 1.7).

The effect of watershed-wide land use on water quality at the outflow of Pinelands impoundments also overshadows that of land use surrounding the lakes (Zampella et al. 2007c). The relationship between altered land and both pH and specific conductance for 30 Pinelands impoundments improved when cumulative lake buffers increased in width, with the watershed-wide land use showing the strongest relationships (Figure 1.8).

Table 1.4. Spearman rank correlations (r) between water-quality and land-use variables for 25 Mullica River Basin stream sites. Water-quality and watershed-wide and distance-weighted altered-land data are the same as those used in Zampella et al. (2007a). Impervious-cover estimates and stream-buffer land-use values were obtained using a GIS and 1995 land-use/land-cover data (NJDEP 2000). Cumulative stream buffers (i.e., 0-100 m, 0-200 m, etc.) encompass the entire length of a stream upstream from a monitoring site. All correlations are significant a $p < 0.001$. For stream buffers > 500 m, n = 24. Conversion: 1 m = 3.28 ft.

Cumulative buffers (m)	pH	Specific conductance	Calcium	Magnesium	Chloride	Nitrate-N
100	0.85	0.87	0.90	0.91	0.86	0.79
200	0.86	0.86	0.90	0.91	0.86	0.78
300	0.88	0.88	0.90	0.92	0.87	0.79
400	0.87	0.87	0.90	0.91	0.87	0.78
500	0.88	0.87	0.90	0.91	0.87	0.79
600	0.87	0.86	0.90	0.91	0.86	0.77
700	0.86	0.87	0.89	0.91	0.87	0.77
800	0.86	0.87	0.90	0.91	0.86	0.76
900	0.86	0.88	0.90	0.91	0.87	0.78
1000	0.86	0.88	0.91	0.92	0.87	0.76
Watershed-wide						
% Altered land	0.90	0.90	0.93	0.93	0.90	0.76
Inverse % altered land	0.89	0.90	0.92	0.94	0.92	0.76
% Impervious surface	0.84	0.86	0.81	0.81	0.94	0.72

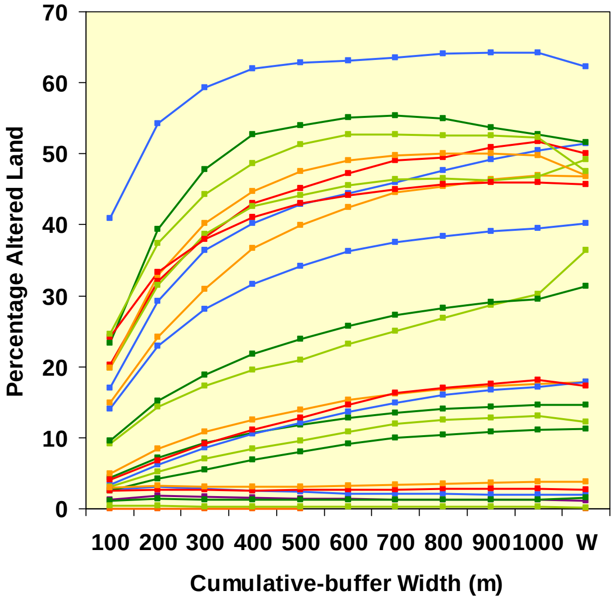

Figure 1.7. The percentage of altered land (developed land and upland agriculture) within cumulative-stream buffers (i.e., 0-100, 0-200, etc.) that encompass the entire length of streams above 25 Mullica River Basin water-quality monitoring sites and the watersheds (W) associated with each site.

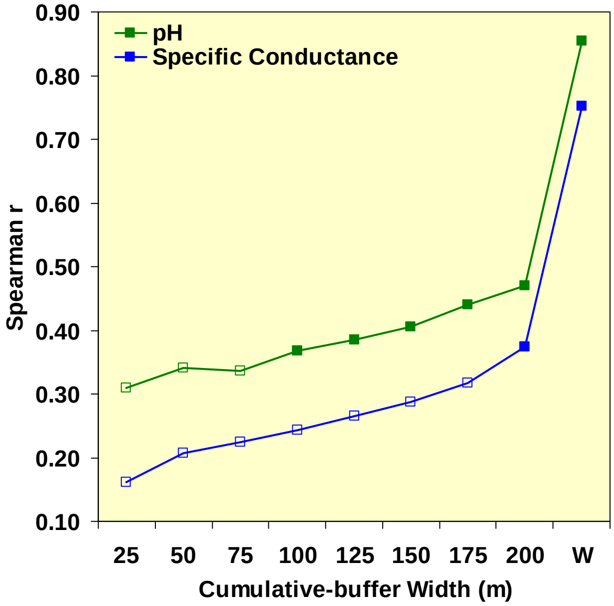

Figure 1.8. Spearman rank correlations (r) relating pH and specific conductance for 30 Pinelands impoundments to the percentage of altered land (developed land an upland agriculture) in the associated watersheds (W) and within cumulative buffers (i.e., 0-25 m, 0-50 m, etc.) surrounding each impoundment. Solid symbols represent relationships significant at *p* $\leq$0.05. From Zampella et al. (2007c).

AQUATIC COMMUNITIES

Human activities continue to degrade aquatic systems throughout North America (Karr and Chu 1999). Many studies have documented an association between the status of stream communities and the extent of developed and agricultural land in a watershed. Sedimentation, changes in stream flows, pesticides, and nutrient enrichment are among the impacts affecting stream biota (Allan 2004). Consistent declines in algal-, invertebrate-, and fish-taxonomic richness are associated with urbanization (Paul and Meyer 2001), whereas the effect of agriculture on aquatic communities is more variable (Allan 2004).

North American Aquatic Studies

Diatoms are generally considered good indicators of water-quality conditions (Meriläinen 1967, Lowe 1974, Charles 1985, van Dam et al. 1994, Pan et al. 1996, Barbour et al. 1999, Battarbee et al. 1999, Stevenson and Pan 1999). Although diatom-taxonomic richness is not consistently related to watershed disturbance (Cuffney et al. 1997, Chessman et al. 1999, Hill et al. 2001, 2003), studies conducted throughout North America have demonstrated an association between variations in the composition of diatom assemblages, water quality, and land use (Leland 1995, Kutka and Richards 1996, Pan et al. 1996, 1999, Carpenter and Waite 2000, Leland and Porter 2000, Hill 2003 et al.). Fewer studies have used vegetation to evaluate the status of aquatic systems (Whitton 1979, Vaithiyanathan and Richardson 1999, Stewart et al. 2003, O'Connor et al. 2000). The majority of aquatic-integrity studies have focused on the composition of stream-macroinvertebrate and fish assemblages.

Macroinvertebrate and fish assemblages. Lenat and Crawford (1994) compared the water quality and aquatic biology of three Piedmont streams in North Carolina. The agricultural and urban watersheds were heavily altered. Row crops covered 48% of the agricultural watershed. Urban land uses represented 69% of the urban basin. Nitrogen and phosphorus concentrations were elevated at both the agricultural and urban sites compared to the forest site and were highest at the agricultural site. Low fish-species richness, low biomass, and the absence of intolerant species characterized the urban-site fish community. Macroinvertebrate assemblages at the agricultural and urban stream sites indicated moderate and severe stress, respectively, based on taxa richness, a Hilsenhoff-type biotic index (Hilsenhoff 1987, which evaluates organic pollution), and the number of species found at only one site. Mayflies were the dominant-macroinvertebrate group at the forested site. Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa richness was lower at the agricultural and urban streams, with the lowest EPT richness found at the urban stream. Oligochaetes were dominant at the urban site and chironomids were dominant at the agricultural sites.

In streams of Georgia's Etowah River basin, urban-land cover was negatively correlated with macroinvertebrate metrics such as total richness, EPT richness, and several different biotic indices that reflected good water quality (Roy et al. 2003). Scott and Hall (1997) compared fish assemblages in Maryland Coastal Plain streams characterized as high-quality and low-quality streams based on a subjective assessment of habitat, water-quality, and land-use characteristics. Fish assemblages in the low-quality, most-impacted streams were dominated by a few tolerant taxa and were less diverse than those in the high-quality, least-impacted streams. Although no land-use data were provided, Scott and Hall (1997) associated impacts affecting the fish assemblages with a range of agricultural- and development-related activities.

Waite and Carpenter (2000) described the composition of fish assemblages in streams representing a range of agricultural land use in Oregon's Willamette Basin. Streams were classified based on species composition. High abundances of tolerant-fish species were collected in small streams with mixed land use and a high percentage of agricultural land. High abundances of introduced species and a high percentage of external abnormalities were found in medium-sized, agricultural-river sites and heavily impacted agricultural-stream sites. The highest abundances of tolerant species were found in the heavily impacted agricultural-stream sites, characterized by lower stream slope, large amounts of fine sediments, high maximum temperatures, and low dissolved-oxygen concentrations. The mean percentage of agricultural land associated with the different stream types ranged from 38 to 62%.

Three studies relating aquatic integrity to land use were conducted in the same Wisconsin streams (Stepenuck et al. 2002, Wang et al. 2000, 2001). Wang et al. (2000) examined the effect of the urbanization of an agricultural region between the 1970s and 1990s on fish assemblages. The analysis included an index of biotic integrity, or IBI (Karr 1981), which is a widely used tool for identifying aquatic degradation based on comparisons of fish assemblages to conditions expected at reference sites. Urbanization was associated with a decrease in the mean number of fish species, fish density, and biotic integrity based on the IBI. For both periods, the numbers of fish species and IBI scores were positively correlated with the percentage of watershed-wide agricultural land and negatively correlated with the percentage of impervious surface. Wang et al. (2000) concluded that although agriculture often degrades stream-fish communities, urban land has a greater impact on a per-unit-area basis.

Wang et al. (2001) associated impervious surface in the Wisconsin stream basins with variations in the number of fish species, fish diversity, fish density, and an IBI. They also suggested that although agriculture can strongly impact stream fishes, degradation of fish communities is not inevitable in predominantly agricultural watersheds. Based on the finding that the percentage of impervious area in the same Wisconsin streams was negatively correlated with macroinvertebrate diversity, the percentage of EPT individuals, and genus richness and positively correlated with the Hilsenhoff biotic index (Hilsenhoff 1987), Stepenuck et al. (2002) concluded that urbanization had a significant impact on stream-macroinvertebrate communities. Agricultural land was positively correlated with diversity and genus richness, which Stepenuck et al. (2002) attributed to the inverse relationship between urban and agricultural land, noting that the results should not be interpreted to mean than agricultural land has a beneficial effect on macroinvertebrate communities.

The effect of scale. Several studies that evaluated the effect of land use on fish and macroinvertebrate assemblages at both watershed-wide and local scales have produced mixed results. Steedman (1988) calculated fish IBI scores for streams in southern Ontario. Mean IBI scores for all samples in each of 10 major and minor river systems were related positively to the proportion of forest in the basin ($r^2 = 0.76$) and negatively to the proportion of urban land use in the basin ($r^2 = -0.64$). When relating IBI scores for 195 individual stream stations to the proportion of urban land in a drainage, Steedman (1988) found that urban land in the area immediately above a stream station explained a greater percentage of the variation in the IBI scores than land use in the entire basin, although the actual difference was generally very small ($r^2 = 0.31$ for the whole basin vs. $r^2 = 0.35$ for the partial basin).

Wang et al. (1997) reported a negative correlation between fish IBI scores for Wisconsin streams and the percentage of agricultural land and urban land in the entire watershed and within 100 m (328 ft) of the streams. IBI scores were positively related to the percentage of forest. An obvious decline in IBI scores was apparent only when agricultural-land use exceeded 50%, whereas a severe decline in IBI scores occurred at low urban-land-use values. Based on their results, Wang et al. (1997) suggested that watershed-wide land-use patterns were generally a better indicator of biological integrity than riparian land uses.

In a study of agricultural streams in eastern Wisconsin, Fitzpatrick et al. (2001) related stream fish, invertebrates, and algal metrics to land use at several scales. A fish IBI was negatively correlated with both the percentage of agriculture within a 50-m (164-ft) stream buffer ($r = -0.76$) and the watershed area outside the buffer ($r = -0.56$). Fish IBI scores considered fair or poor were obtained when watershed-wide agriculture was above 30%. All but one stream received poor IBI scores when agriculture within the buffer exceeded 10%. A diatom IBI was correlated with watershed-wide agriculture (r = -0.40), but not with the percentage of agriculture in the 50-m buffer. A family-level Hilsenhoff-type macroinvertebrate biotic index (Hilsenhoff 1988) was not related to agriculture at either scale.

Lammert and Allan (1999), who sampled fish and macroinvertebrates in tributaries of an agricultural watershed in Michigan, found that land use within 100 m (328 ft) of a stream was related to several measures of biological condition, including a fish IBI, whereas land use in the entire watershed was unrelated to biological conditions. In contrast, Roth et al. (1996) sampled fish in the same Michigan watershed and found that basin-wide land use was a better predictor of stream-biotic integrity, based on an IBI, than riparian conditions at the local (150-m or 492-ft stream length) and reach (1,500-m or 4,921-ft stream length) scale. In the Roth et al. (1996) study, biotic integrity was negatively associated with the percentage of agriculture in a basin, which explained 49.6% of the variation in the IBI, and the percentage of agriculture within 50-m of a stream, which explained 37.8% of the variation.

King et al. (2005) indicated that as little as 21% developed land in a watershed may result in a change in the composition of macroinvertebrate assemblages in Maryland Coastal Plain streams, with a high probability that sharp changes in composition would occur in watersheds with more than 32% developed land. Based on results using inverse-distance-weighted percentage developed land, they suggested that both local and watershed-wide development influenced stream conditions, with development near the stream station having a greater effect. Likewise, macroinvertebrate richness, diversity, and the number of EPT taxa in Appalachian headwater streams were most closely related to the percentage of agricultural and urban land within a 30-m (98-ft) wide riparian buffer extending 200 m (656 ft) upstream from the sampling sites compared with basin-wide land use (Sponseller et al. 2001).

In West Virginia, Snyder et al. (2003) used two different IBIs to evaluate the relationship between stream-fish assemblages and land use at the watershed and riparian-buffer scales. A negative relationship was found between both IBIs and the extent of watershed-wide urban land. One of the two IBIs was positively correlated with watershed-wide agricultural land, which they attributed to the inverse relationship between urban land and agriculture and the extent and type of agriculture. Agricultural land was correlated with nitrate concentrations and the proportion of fine sediments in riffles. Neither index was correlated with riparian-scale land use. Snyder et al. (2003) reached the same conclusion as Wang et al. (2000), indicating that on a per-unit-area basis, urban land has a greater impact on streams than agriculture.

Morley and Karr (2002) used a 10-metric macroinvertebrate IBI to assess stream health in relation to urban-land cover in the Puget Sound Basin of western Washington. Biological integrity decreased as the percentage of urban-land cover increased. The correlation between urban land and most of the 10 macroinvertebrate-based metrics included in the IBI was stronger for upstream land use in the entire drainage area than for land use within a 200-m (656-ft) buffer extending 1 km upstream from a sampling point. However, the percentage of variation in the overall IBI explained by basin-wide, riparian, and local land use was similar.

Moore and Palmer (2005), who studied small headwater streams in the Piedmont region of Maryland, reported higher macroinvertebrate richness and diversity in agricultural streams compared to urban streams and agricultural streams in other regions. They attributed the higher biodiversity in the Maryland agricultural streams to the widespread use of best management practices such as no-till farming and the use of riparian buffers. An increase in biodiversity in urban streams was associated with the amount of riparian forest within 30 m (98 ft) of a stream. Allan (2004) suggested that the greater influence attributed to proximate land uses in some studies may be due to the greater variation in land cover in riparian areas compared to the variation within a watershed.

Amphibians. Although amphibians may be good indicators of environmental conditions (Hecnar and M'Closkey 1996, Wake 1991), amphibian assemblages have not been used as frequently as other taxonomic groups to assess biotic integrity. Simon et al. (2000) developed an IBI using crayfish,

fish, and amphibians to assess vernal ponds and small palustrine wetlands along the shore of Lake Michigan. Hughes et al. (2004) related an IBI for fish and amphibian assemblages in coldwater streams of Oregon and Washington in relation to natural gradients and human-related disturbances, where watershed disturbance was based on road density and vegetation cover. Crewe and Timmermans (2005) developed and applied an IBI using amphibian assemblages to assess the integrity of Great Lakes coastal wetlands in relation to a disturbance gradient characterized by percentage cover of marsh, woodland, urban land, and agriculture in surrounding areas. All metrics, including total species richness and abundance, were negatively correlated with land-use disturbance. Moyle and Randall (1998) included a metric based on the presence and abundance of native ranid frogs and several fish-based metrics in an IBI for Sierra Nevada watersheds in California. Sierra Nevada watersheds that have been highly altered by dams, urbanization, agriculture, and mining and watersheds with introduced fish and frogs were among those receiving low IBI scores. A comparison of buffered (fenced from cattle access) and unbuffered streams in a southwestern Georgia agricultural landscape revealed no differences in amphibian presence or abundance within riparian areas, although in-stream larval salamanders were more abundant and amphibian-species richness tended to be higher in buffered streams (Muenz et al. 2006).

In a Canadian study, Bishop et al. (1999) suggested that nutrient runoff was associated with lower anuran-species diversity and density at a muck-soil-agriculture area compared to upstream and downstream wetland areas. Welsh and Ollivier (1998) reported that amphibian densities in California streams impacted by sedimentation from road construction were lower than densities in unimpacted streams. In another Canadian study, de Solla et al. (2002) found that the hatching success of northern red-legged frogs (*Rana aurora*) and northwestern salamanders (*Ambystoma gracile*) was lower in roadside ditches at agricultural sites compared to reference sites, although they observed no difference in hatching success for laboratory-reared eggs of northern red-legged frogs exposed to water samples taken from the field sites.

Nitrogen concentrations have been associated with lethal and sublethal effects on amphibians, but results vary among different studies. Rouse et al. (1999) concluded that nitrate concentrations in some North American watersheds are high enough to cause lethal and sublethal effects in amphibians. Knutson et al. (2004) observed negative effects on anuran reproduction in ponds at total nitrogen concentrations ranging from 0.1 to 14 mg L^{-1}. Both species richness and reproductive success decreased with increasing concentrations of total nitrogen.

In laboratory experiments, Marco et al. (1999) exposed larvae of five Pacific Northwest amphibian species, including the Oregon spotted frog (*Rana pretiosa*), red-legged frog, western toad (*Bufo boreas*), Pacific treefrog (*Hyla regilla*), and northwestern salamander, to a range of nitrite (0-25 mg L^{-1}) and nitrate (0-7 mg L^{-1}) concentrations for a period of 15 days. They found that, compared to control concentrations, some larvae of some species reduced feeding activity, swam less vigorously, showed disequilibrium and paralysis, suffered abnormalities and edemas, and eventually died in the presence of either nitrite or nitrate ions. Hecnar (1995) conducted acute and chronic laboratory experiments by exposing larvae of the American toad, western chorus frog (*Pseudacris t. triseriata*), northern leopard frog, and green frog to ammonium-nitrate fertilizer. In the acute exposures, which ranged from 0-50 mg L^{-1} nitrate, mortality varied among species and reduced activity, weight loss, and physical abnormalities were observed for all species. During chronic 100-day-long exposures to nitrate concentrations that ranged from 0-10 mg L^{-1}, chorus frog and leopard frog tadpoles showed lower survivorship in the 10 mg L^{-1} treatment, but green frog survivorship was unaffected by nitrate concentration. In another laboratory experiment, Laposata and Dunson (1998) exposed wood frog, Jefferson salamander, spotted salamander, and American toad eggs to nitrate concentrations ranging from 0 to 40 mg L^{-1} and found no difference in the hatching success or proportion of deformed larvae among the treatments.

Allran and Karasov (2000) tested the effects of atrazine and nitrate on northern leopard frogs in the laboratory by exposing larvae to varying concentrations of both atrazine, ranging from 0 to 200 μg L^{-1}, and nitrate nitrogen, ranging from 0 to 30 mg L^{-1}. Although nitrate slowed the growth of the larvae, Allran and Karasov (2000) concluded that concentrations of atrazine and nitrate commonly found in the environment are not likely to affect northern leopard frog larvae through direct toxicity.

Eutrophic conditions characterized by elevated pH, water temperature, and un-ionized ammonia (NH_3) may also be associated with frog-embryo mortality or malformations (Boyer and Grue 1995), although Jofre and Karasov (1999), who exposed anuran embryos to varying concentrations of un-ionized ammonia, ranging from 0 to 2 mg L^{-1}, suggested that anurans may not be as sensitive to ammonia as many fish species. In their study, survival and development of green frogs and leopard frogs were affected at concentrations of 0.6 mg L^{-1} and 1.5 mg L^{-1}, respectively, and the prevalence of deformities increased at those levels. American toads did not show any effects up to a concentration of 0.9 mg L^{-1} NH_3.

Pinelands Aquatic Studies

Macroinvertebrates. Using a rapid-bioassessment approach, Kennen (1999) related stream-macroinvertebrate-community impairment to watershed characteristics in New Jersey streams. Urban land and municipal-effluent discharges were positively associated with severe macroinvertebrate-community impairment, whereas the extent of forested land was inversely related to severe impairment. Severely impaired-macroinvertebrate communities were more likely to occur in regions of New Jersey containing urban centers compared to regions located in the less urbanized northwest portion of the state. The Coastal Plain region, which includes the Pinelands, had the highest probability of exhibiting an impaired-macroinvertebrate community. Kennen (1999) suggested that the low bioassessment scores for this region might be due to the naturally depauperate communities in Pinelands streams. Jessup et al. (2005) supported this conclusion, indicating that methods used to assess impairment in other parts of New Jersey are inadequate in the Pinelands.

Because chironomids are relatively tolerant of organic pollution (Hilsenhoff 1987, 1988), they are often used as an indicator of aquatic degradation. For example, Jones and Clark (1987) reported that the relative abundance of two chironomid genera increased in more highly urbanized-stream sites in northern Virginia. In North Carolina Piedmont streams, chironomids were more abundant in nutrient-enriched agricultural streams compared to forest streams. However, in the Pinelands, chironomids are frequently among the dominant macroinvertebrates in the least-disturbed aquatic habitats (Dougherty and Morgan 1991, Zampella et al. 2008). Chironomids have also been shown to be an important taxonomic group in other Coastal Plain streams (Smith and Smock 1992, Wright and Smock 2001).

Many macroinvertebrates found in Pinelands streams are adapted to lentic habitats and slow-moving lotic habitats or low-oxygen environments, characteristics that are generally associated with aquatic impairment in other regions, but that are typical in the low-gradient Pinelands streams (Zampella et al. 2008). Fingernail clams (Sphaeriidae = Pisidiidae) are among the few mollusks found in small, acidic Coastal Plain streams due most likely to a greater tolerance to high acidity (Smock and Gilinsky 1992, Pennak 1978). Dougherty and Morgan (1991), who compared macroinvertebrates in Nescochague Lake and Oswego Lake, collected mollusks (fingernail clams) and flatworms (*Planaria*) only in Nescochague Lake, a nutrient-enriched Pinelands impoundment. Jessup et al. (2005) listed the percentage of mollusks and amphipods among the indicators of biological degradation in Pinelands streams. Analysis of macroinvertebrate data for 50 Mullica River Basin stream sites from Poretti et al. (2001) and land-use and water-quality data for the same sites from Zampella et al. (2001) reveals that the distribution of two mollusk taxa (fingernail clams and planorbid snails) is positively associated with the percentage of altered land, consisting of developed land and upland agriculture, in a basin, pH, and specific conductance (Figure 1.9). Both pH and specific conductance are positively correlated with calcium concentrations in Mullica River Basin streams (Zampella et al. 2001, 2007a). Calcium is a major constituent in the shells of mollusks.

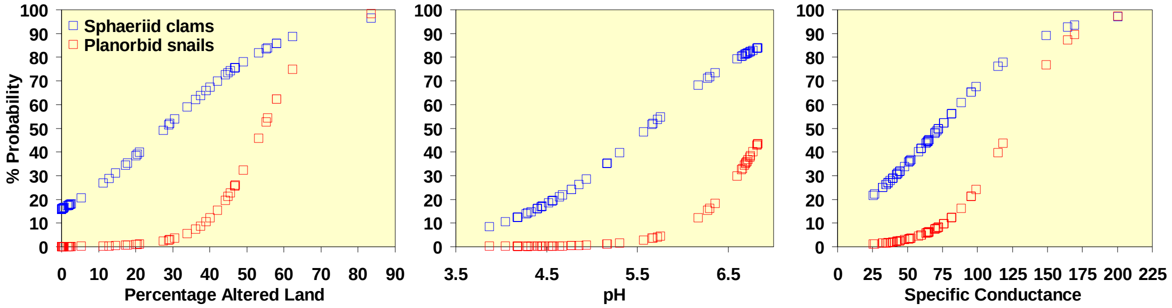

Figure 1.9. The probability of encountering sphaeriid clams and planorbid snails in relation to the percentage of altered land in a watershed and stream pH and specific conductance (n = 50). Source of data: Poretti et al. (2001) and Zampella et al. (2001).

Although stream-macroinvertebrate composition may be related to present-day land uses, the legacy of historic land uses, especially agriculture, must also be considered (Foster et al. 2003, Harding et al. 1998). Bennett et al. (2004) described the macroinvertebrate communities in streams impacted by urban land, pasture, and row crop as impaired when compared to least impacted streams in the Choctawhatchee-Pea River watershed in southeast Alabama, a region characterized by low elevation, sandy soils, and forests dominated by pines, oaks, and other mixed hardwoods. The investigators concluded that macroinvertebrate communities in the watershed reflect the history of agricultural and silvicultural practices. In the Pinelands, the composition of stream-macroinvertebrate assemblages associated with present-day cranberry agriculture differed from those found in reference streams draining undisturbed forest (Zampella et al. 2008). The composition of assemblages in streams draining abandoned bogs shared similarities with both cranberry streams and forest streams.

Zooplankton. Zooplankton have rarely been used to evaluate ecological integrity (Lougheed and Chow-Frasera 2002). Dodson et al. (2005) observed that zooplankton diversity in shallow Wisconsin lakes was correlated with agricultural land-use within a 30-m (98-ft) riparian zone. Agricultural-land use was associated with little or no riparian and aquatic vegetation. In the only zooplankton study conducted in the Pinelands, Morgan (1985) found that zooplankton composition was similar in dystrophic Oswego Lake and nutrient-enriched Nescochague Lake.

Fish. The introduction and establishment of nonnative fish is associated with human-related degradation of aquatic habitats throughout North America (Moyle 1986). Based on qualitative descriptions of land-use patterns in the Mullica River Basin, Hastings (1984) associated the presence of nonnative fishes, including species found in areas peripheral to the Pinelands and those introduced to New Jersey, with Pinelands streams affected by development and agriculture. Hastings (1984) indicated that acidity seemed to be the major factor limiting the distribution of many fish species. Graham (1993) found that species richness in New Jersey lakes decreased with increasing acidity due to a decline in the numbers of exotic-, estuarine-, and anadromous-fish species and that the probability of occurrence of species that are not native to New Jersey, such as largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and black crappie (*Pomoxis nigromaculatus*), increased as lake pH increased. Graham and Hastings (1984) suggested that the relative scarcity of bluegills and pumpkinseeds in the more acidic waters of the Pinelands was due to the absence of suitable food for young fish in dystrophic waters, which are characterized by high humic color and low phytoplankton production (Wetzel 2001), rather than the direct physiological effect of low pH. Regardless of the mechanism, it appears that Pinelands waters with circumneutral pH offer less environmental resistance to the successful establishment of nonnative fishes than the region's more-acid waters (Zampella and Bunnell 1998).

Following on the work of Hastings (1984), Zampella and Bunnell (1998) quantified the relationship between the watershed disturbance and stream-fish-species composition in the Mullica River Basin. A fish-community gradient, characterized by an increase in the percentage of nonnative fish species, paralleled an environmental gradient, represented by an increase in the percentage of developed and agricultural land, pH, and specific conductance. Nonnative fish were present only at stream sites with elevated pH and specific conductance values and a high percentage of altered land in the watershed. Subsequent fish-surveys conducted in streams and impoundments throughout the Pinelands have revealed similar patterns (Table 1.5, Zampella et al. 2001, 2003, 2005, 2006a, 2007c). The lack of strong or consistent relationships between fish-community gradients and watershed-disturbance variables in the Great Egg Harbor River Watershed Management Area (Table 1.5) may be due to the general absence of minimally disturbed survey sites and the widespread distribution of nonnative species (Zampella et al. 2005). The presence of nonnative centrarchids (sunfishes) in degraded Pinelands waters may pose a special problem because some species have been associated with a decrease in native-fish-species richness, a loss of native-frog species, and the invasion of bullfrogs (Hayes and Jennings 1986, Kiesecker and Blaustein 1998, Chapleau et al. 1997, Whittier et al. 1997, Adams 1999, Findlay et al. 2000, Adams et al. 2003). The results of Pinelands fish surveys indicate that wetland agriculture is not associated with the presence of nonnative fish species, a finding that is supported by a study conducted by Bunnell (2006), who found that only native fish occurred at Pinelands streams draining forest, abandoned-cranberry bogs, and active-cranberry bogs.

Anurans. As with Pinelands fish communities, variations in the region's anuran (frog and toad) assemblages are usually associated with watershed disturbance represented by an increase in the percentage of altered land (developed land and upland agriculture), pH, and specific conductance (Bunnell and Zampella 1999, Zampella and Bunnell 2000, Table 1.5). Assemblages with border-entrant anurans are generally found at degraded sites, whereas assemblages composed of native Pine Barrens species and wide-ranging species are found at sites in relatively unaltered watersheds. As previously described, border entrants are species, such as the bullfrog, that are widely distributed outside the Pinelands but usually do not occur in the region except in habitats altered by human activity. Pine Barrens species, which include Pine Barrens treefrogs and carpenter frogs, are restricted to Pinelands habitats, and wide-ranging species are distributed throughout southern New Jersey (Conant 1962, 1979).

Although the anuran-community gradient in the Great Egg Harbor River Watershed Management Area was not associated with watershed-disturbance variables (Table 1.5), the percentage of native species at a site decreased and the percentage of nonnative species at a site increased as pH increased. Also, as the percentage of altered land in a basin increased, the number of native carpenter frogs heard at a site decreased and the number of nonnative bullfrogs heard increased. Throughout the Pinelands, the probability of encountering carpenter frogs decreases as the percentage of altered land in a basin increases (Zampella and Bunnell 2000, Figure 1.10). The opposite trend is found for bullfrogs. The reasons for the lack of strong or consistent relationships between the anuran-community gradient and watershed-disturbance variables in the Great Egg Harbor River watershed are similar to those given for fish, that is, minimally disturbed survey sites were generally absent and nonnative species were widespread (Zampella et al. 2005).

Stream vegetation. Morgan and Philipp (1986) compared the water quality and plant-species composition of six central Pinelands-stream sites that they characterized as either polluted or unpolluted. Lands in the headwater areas of the polluted streams were described as extensively developed or farmed. Upstream regions of unpolluted sites were described as undeveloped. Nitrate-nitrogen concentrations and pH values were significantly higher in the polluted streams. The polluted and unpolluted streams each supported a unique set of plant species. Some typical Pinelands species were replaced by species that are characteristic of areas adjacent to the Pinelands or other parts of the United States.

Table 1.5. Spearman rank correlation coefficients (r) relating altered land, pH, and specific conductance to biological-community gradients represented by the first axis of detrended correspondence analysis (DCA) ordinations. Altered land represents the combined percentage of developed land and upland agriculture in a watershed. The four community gradients are characterized by an increase in the percentage of non-Pinelands species. From Zampella et al. (2006a).

Watershed	Stream vegetation			Anurans			Stream fish			Impoundment fish		
Variable	n	r	*p*	n	r	*p*	n	r	*p*	n	r	*p*
Mullica River												
Altered land	72	0.79	<0.001	78	0.77	<0.001	54	0.82	<0.001	30	0.90	<0.001
pH	72	0.73	<0.001	41	0.77	<0.001	54	0.82	<0.001	30	0.89	0.000
Specific conductance	72	0.68	<0.001	41	0.61	<0.001	54	0.62	<0.001	30	0.41	0.024
Rancocas Creek												
Altered land	44	0.75	<0.001	42	0.71	<0.001	41	0.75	<0.001	15	0.50	0.057
pH	44	0.66	<0.001	29	0.74	<0.001	41	0.87	<0.001	11	0.65	0.029
Specific conductance	44	0.70	<0.001	29	0.62	<0.001	41	0.63	<0.001	11	0.32	0.332
Great Egg Harbor River												
Altered land	36	0.61	<0.001	27	0.15	0.456	32	0.31	0.081	10	0.66	0.038
pH	36	0.73	<0.001	23	0.28	0.197	32	0.36	0.046	10	0.64	0.048
Specific conductance	36	0.33	<0.001	23	0.11	0.611	32	0.05	0.806	10	0.01	0.987
Barnegat Bay												
Altered land	50	0.61	<0.001	34	0.64	<0.001	40	0.55	<0.001	17	0.56	0.019
pH	50	0.46	<0.001	19	0.68	0.001	40	0.64	<0.001	16	0.56	0.025
Specific conductance	50	0.54	<0.001	19	0.00	1.000	40	0.46	0.003	16	0.67	0.005

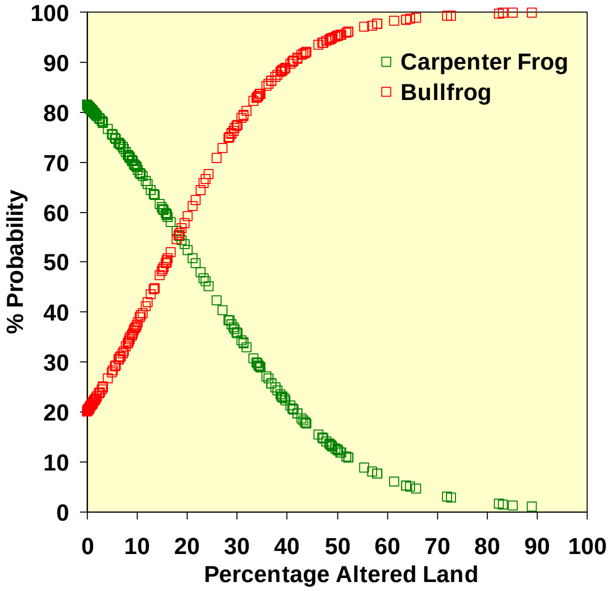

Figure 1.10. The probability of encountering carpenter frogs and bullfrogs in relation to the percentage of altered land in a watershed (n = 181 sites). Source of data: Zampella et al. (2001, 2003, 2005, 2006a).

Zampella and Laidig (1997) related the composition of stream-channel and stream-bank plant-species in the Mullica River Basin to a complex watershed-disturbance gradient characterized by an increase in the extent of upland agriculture and developed land, pH, specific conductance, and channel muck. Surface waters in relatively undisturbed basins displayed lower pH and specific conductance values compared with waters in highly developed or farmed watersheds. Sites within highly altered basins supported a unique group of non-Pinelands plant species, referred to as disturbance-indicator species. Subsequent surveys conducted in each of the four major Pinelands watersheds confirmed the relationship between the presence of non-Pinelands plant species and the percentage of developed land and upland agriculture in a watershed (Table 1.5, Zampella et al. 2001, 2003, 2005, 2006a, 2007c). The results of Pinelands stream-vegetation studies indicate that wetland agriculture is not associated with the presence of nonnative-plant species, a finding that is supported by a study conducted by Laidig (2006) who compared channel and bank vegetation in 11 Pinelands streams draining forest, abandoned-cranberry bogs, and active-cranberry bogs.

Diatoms. Acid-water species characterize the Pinelands-diatom flora (Patrick et al. 1979, Moul and Buell 1979, Patrick 1996). Three Pinelands studies have examined the relationship between watershed disturbance and the composition of diatoms. Morgan (1987) collected attached algae, including diatoms, from three undisturbed Pinelands streams, characterized by undeveloped forest, and three disturbed Pinelands streams, characterized by elevated pH, nutrients, and a higher percentage of developed land and upland agriculture than the undisturbed-forested basins. Morgan (1987) identified nine diatom species as restricted to disturbed sites and nine species as restricted to undisturbed sites. He concluded that disturbance resulted in the replacement of characteristic Pinelands periphyton species by species peripheral to the region. He also concluded that periphyton-taxonomic richness increased with disturbance.

Zampella et al. (2007b) compared the composition of diatom assemblages collected from Pinelands streams draining forest, abandoned-cranberry bogs, active-cranberry bogs, and altered land consisting of developed land and upland agriculture. Altered land, which ranged from 29% to 62% in the developed and agricultural basins, was less than 10% in the other stream basins. Differences in diatom-species composition were associated with variations in pH and specific conductance. Diatoms considered to be most widely distributed at pH values below 7 by Meriläinen (1967), Lowe (1974), van Dam et al. (1994), or Charles (1985) and classified by Zampella et al. (2007b) as acidobiontic-acidophilous characterized the dominant species at forest, abandoned-bog, and cranberry sites, whereas species that are generally associated with a pH of around 7 and classified as indifferent, dominated the developed- and agricultural-site samples. In a separate study, Zampella et al. (2007c) sampled diatoms at the outflow of Pinelands lakes. As in the earlier study, the percentage of dominant species classified as acidobiontic-acidophilous diatoms decreased and those classified as indifferent increased as the percentage of altered land in a watershed increased.

Although Zampella et al. (2007b) identified several species that distinguished developed and agricultural streams from the cranberry and forest streams based on differences in abundance and frequency of occurrence, none of these species was unique to developed and agricultural, cranberry, or forest streams. In contrast to the Morgan (1987) study, Zampella et al. (2007b) found that although diatom species and genus richness was generally higher in streams draining altered land compared to streams draining cranberry bogs, abandoned bogs, or forest, the differences were not statistically significant.

Multiple indicators. Several studies have used multiple indicators to assess the ecological integrity of aquatic systems (Berkman et al. 1986, Moyle and Randall 1998, Allen et al. 1999, Simon et al. 2000, Fitzpatrick et al. 2001, Stewart et al. 2003, Yoder and DeShon 2003, Wang and Lyons 2003). Because different biological indicators respond to human disturbances at different scales (Karr and Chu 1999), the use of an ecological indicator that represents a composite of several single indicators may provide a more comprehensive evaluation of ecological integrity than one based on single indicators. Zampella et al. (2006b) used a multiple-indicator approach to characterize the ecological integrity of Mullica River Basin streams. With the exception of impoundment fish, the association between altered land and multiple-indicator scores based on pH, specific conductance, stream vegetation, and stream-fish, impoundment-fish, and anuran assemblages was stronger than that displayed by any of the individual indicators.

Proximity and the relative effect of agriculture and developed land. As previously mentioned, North American studies evaluating the effect of land use on fish and macroinvertebrate assemblages at both watershed-wide and local scales have produced mixed results, but accounting for proximity generally produced results similar to those obtained using the simple proportion of developed or agricultural land in a watershed. Also, although agriculture has been associated with impaired aquatic communities, it has been suggested that in some cases urban land has a greater impact.

As with water quality, the effect of watershed-wide land use on Pinelands impoundment communities also appears to overshadow that of the land use surrounding the lakes. The relationship between altered land and the percentage of nonnative plant and fish species improves dramatically when cumulative lake buffers increase in width, with the strongest relationships associated with watershed-wide land use (Figure 1.11, Zampella et al. 2007c).

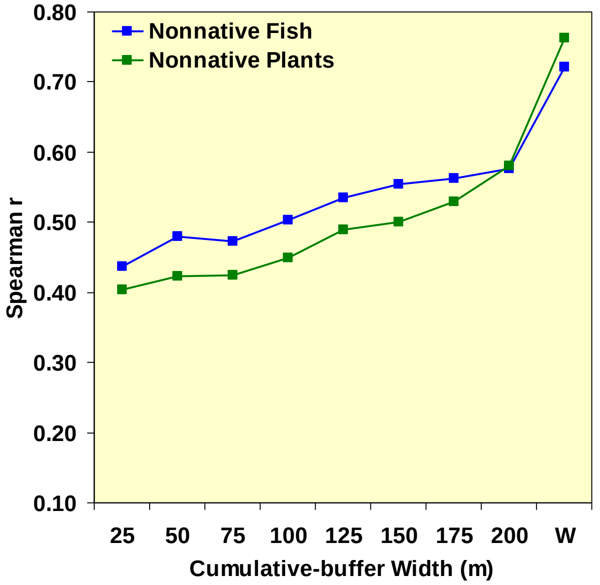

Figure 1.11. Spearman rank correlations (r) relating the percentage of nonnative fish and plants in 30 Pinelands impoundments to the percentage of altered land (developed land and upland agriculture) in the associated watersheds and within cumulative buffers (i.e., 0-25 m, 0-50 m, etc.) surrounding each impoundment. All relationships are significant at $p \leq 0.05$. From Zampella et al. (2007c). Conversion: 1 m = 3.28 ft.

Zampella et al. (2002) used multiple regression and community-composition scores, represented by first-axis detrended correspondence analysis (DCA) sample scores for 72 stream sites in the Mullica River Basin, to determine the percentage of variation in plant-species composition explained by developed land and upland agriculture in associated watersheds. Higher community-composition scores were associated with sites characterized by a higher percentage of non-Pinelands plant species. Upland agriculture and developed land, which were positively correlated with the community-composition scores, explained 68% of the variation in plant-species composition, with upland agriculture alone accounting for 58% of the variation. Including the proximity of developed and agricultural land to a stream-monitoring site using inverse-distance-weighted measures did not improve the relationship between land use and stream vegetation derived using unweighted land-use data. The same approach was used to relate fish-species composition at 54 survey sites to land use. Upland agriculture and developed land were positively correlated with the community-composition scores in all models. Higher community-composition scores were associated with sites characterized by a higher percentage of nonnative fish species. The distance-weighted model explained slightly more of the variation in fish-community composition (74%) compared to the model based on unweighted-land-use data (69%). Upland agriculture accounted for 56% and 57% of the variation in the unweighted and distance-weighted models, respectively.

The relative effect of developed land and upland agriculture on Pinelands stream communities was explored further by analyzing data from watersheds where each land-use type represented at least two-thirds of the altered land in watersheds with at least 1% altered land. Analysis of data presented in Zampella et al. (2001, 2003, 2005, 2006a) revealed that the percentage of nonnative fish and plants in Pinelands streams is positively related to the percentage of altered land in watersheds regardless of whether upland agriculture or developed land is the dominant altered-land-use type (Tables 1.6 and 1.7). The relationship between altered land and nonnative-stream fish is stronger for 24 basins dominated by agriculture compared to 82 basins dominated by developed land. Logistic regression shows that below about 25% altered land, the probability of encountering a nonnative-fish species is greater in watersheds dominated by developed land compared to those dominated by agricultural land (Figure 1.12). When altered land exceeds 25%, the probability of encountering nonnative fish becomes greater in watersheds dominated by agriculture.

The relationship between altered land and non-Pinelands plants is also stronger for 30 basins dominated by agriculture compared to 90 basins dominated by developed land (Tables 1.6 and 1.7). Logistic regression shows that below about 10% altered land, the probability of encountering stream vegetation composed of at least 20% non-Pinelands plants is greater in watersheds dominated by developed land than those dominated by upland agricultural land (Figure 1.12). However, in basins with more than 10% altered land the probability of encountering sites where at least 20% of the species are non-Pinelands plants increases dramatically in the watersheds dominated by agriculture.

WETLAND HYDROLOGY AND WATER QUALITY

Wetland Hydrology

Hydrology is a major determinant of wetland-vegetation patterns (Mitsch and Gosselink 2000). In the Pinelands, relatively distinct forest communities are associated with narrow ranges of water levels (Figure 1.13). Transitional pine forests on mineral soils dominate the drier end of the wetland-forest gradient, whereas hardwood and Atlantic white cedar swamps underlain by organic soils occupy the wetter end of the hydrologic continuum (Zampella et al. 1992, Zampella 1994, Laidig and Zampella 1999).

Wetland hydrology is also a key factor that directly shapes the community composition of aquatic-breeding amphibians. The timing and duration (hydroperiod) of wetland inundation must coincide with the breeding phenology and larval ontogeny of an amphibian species for successful mating, oviposition, hatching, and metamorphosis to occur (Semlitsch et al. 1996, Paton and Crouch 2002). Because amphibian species differ in their hydrologic requirements, amphibian-community composition can vary along temporary-to-permanent water hydroperiod gradients. For example, in South Carolina, amphibians displayed a unimodal response to inundation where intermediate hydroperiod (8-10 months) wetlands supported the highest larval-amphibian-species richness (Snodgrass et al. 2000). In New Hampshire, larval-amphibian richness was found to be lowest in short-hydroperiod wetlands (<4 months) compared to intermediate (4-11 months) and permanent wetlands (Babbitt 2005).

Table 1.6. Spearman rank correlations between the percentage of nonnative fish and the percentage of developed land, upland agriculture, and altered land (developed land and upland agriculture) in Pinelands watersheds dominated by agriculture or developed land. The dominant land use represented at least two-thirds of the altered land in a watershed. Neither developed nor agricultural land was dominant at 61 sites.

	% Nonnative fish at sites dominated by developed land			% Nonnative fish at sites dominated by upland agriculture			% Nonnative fish at all sites		
	n	r	p	n	r	p	n	r	P
% Developed land	82	0.60	<0.001	24	0.74	<0.001	167	0.64	<0.001
% Upland agriculture	82	0.26	0.021	24	0.72	<0.001	167	0.30	<0.001
% Altered land	82	0.56	<0.001	24	0.73	<0.001	167	0.65	<0.001

Table 1.7. Spearman rank correlations between the percentage of non-Pinelands plants and the percentage of developed land, upland agriculture, and altered land (developed land and upland agriculture) in Pinelands watersheds dominated by agriculture or developed land. The dominant land use comprised at least two-thirds of the altered land in a watershed. Neither developed nor agricultural land was dominant at 82 sites.

	% Non-Pinelands plants at sites dominated by developed land			% Non-Pinelands plants at sites dominated by upland agriculture			% Non-Pinelands plants at all sites		
	n	r	p	n	r	p	n	r	p
% Developed land	90	0.57	<0.001	30	0.76	<0.001	202	0.64	<0.001
% Upland agriculture	90	0.54	<0.001	30	0.76	<0.001	202	0.51	<0.001
% Altered land	90	0.58	<0.001	30	0.75	<0.001	202	0.68	<0.001

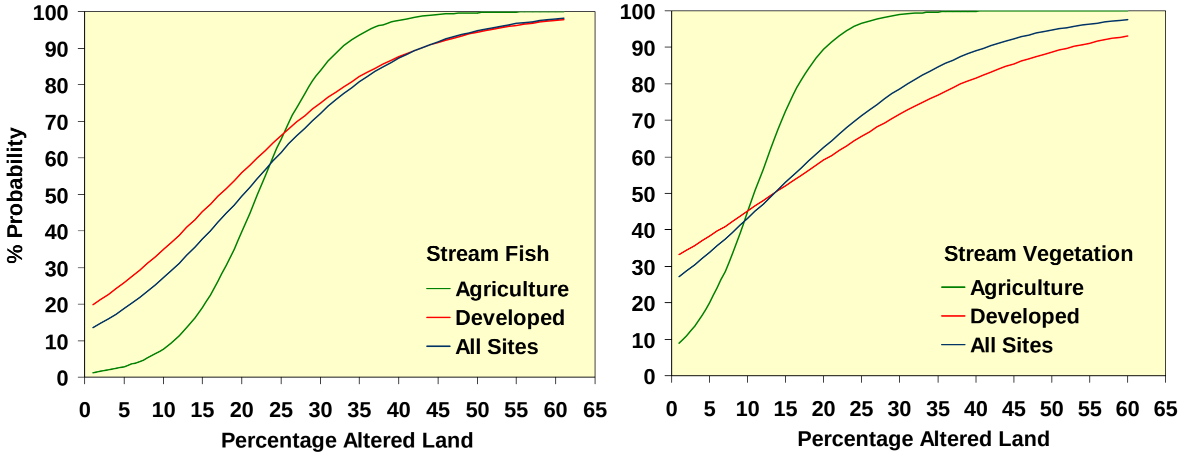

Figure 1.12. The probability of encountering fish assemblages with nonnative species or vegetation composed of at least 20% nonnative plants at stream sites in watersheds dominated by either developed land or upland agriculture in relation to the percentage of altered land (developed land and upland agriculture) in a watershed. Source of data: Zampella et al. (2001, 2003, 2005, 2006a).

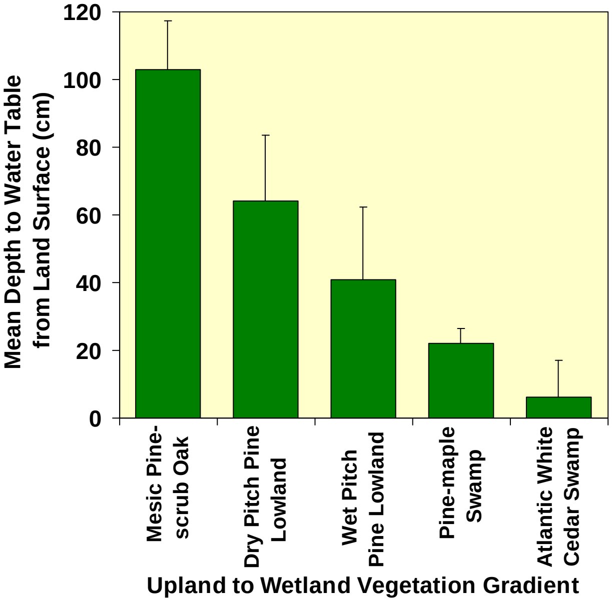

Figure 1.13. Mean 2-year water-table level (± 1 standard deviation), expressed as the distance below the land surface, for five Pinelands vegetation types. Based on data from Zampella et al. 1992 and Laidig and Zampella 1999.

Long-term field studies have documented a strong link between pond hydroperiod and amphibian-recruitment success. In a sixteen-year study of a natural Carolina bay, the number of individuals and the number of species that metamorphosed increased with hydroperiod (Semlitsch et al. 1996). Similar relationships were obtained when a subset of data from the same bay were combined with a four-year dataset from a ditched Carolina bay and a three-year dataset from an artificial wetland (borrow pit) (Pechmann et al. 1989). Experimental manipulations in replicated artificial-pond habitats have also shown that hydroperiod can effect larval period, survival, and the number of metamorphs (Rowe and Dunson 1995, Ryan and Winne 2001). Wetland hydrology can also indirectly influence amphibian communities by regulating populations of aquatic predators, such as invertebrates, salamanders, and fish, which tend to become more abundant as hydroperiod increases (Woodward 1983, Skelly 1996).

The hydrologic system supporting wetlands is a continuum in which any modification will impact contiguous parts of the system (Winter 1988). Even isolated wetlands are connected to streams via groundwater-flow systems (Winter 2003). Although hydrology is one of the primary factors influencing the structure and function of wetlands, few studies have documented the effect of altered hydrologic regimes or nutrient enrichment on wetland habitats. Determining wetland and groundwater interactions is complicated by the difficulty of defining flow-system boundaries, flow paths, recharge and discharge, hydraulic gradients, and aquifer properties (Winter 1981, 1988, Winter and LaBaugh 2003). This difficulty is a major reason for the general lack of empirical evidence relating land-use impacts to variations in the ecological integrity of wetland habitats.

In the Pinelands, most of the Kirkwood-Cohansey groundwater used by a home or business with a private well and a septic system is returned to the local-groundwater system, whereas developments served by a regional-sewer system or supplied by distant public wells deplete local groundwater sources. Agriculture also removes water from the Kirkwood-Cohansey aquifer system, although consumptive use varies among different crops (Clawges and Titus 1993). In an unconfined aquifer with permeable sediments, groundwater pumping results in a cone of depression that is generally shallow and extensive

(Winter 1988). The effect of pumping will directly impact a wetland if the cone of depression intersects it. Even if the cone of depression does not intersect a wetland, a well can decrease discharge to a wetland system. Because of the integral relationship between groundwater and surface water in the Pinelands, it is generally assumed that pumping will affect water levels in wetlands and stream discharge even if the pumping center is a distance from the wetland. The assumption has a sound theoretical basis that has been demonstrated with modeling, but the empirical evidence needed to quantify the relationship is generally lacking. The potential for such impacts was demonstrated by a pump test along the Mullica River in Wharton State Forest (Lang and Rhodehamel 1963). After about six days of pumping, water levels declined in swamps located on both sides of the river.

Faulkner (2004) suggested that urbanization has a greater impact on wetlands than agriculture due to the dissimilarity between urban and natural areas and to the permanent nature of urban lands and associated hydrologic changes. Urbanization in the Puget Sound region of Washington has been associated with an increase in wetland water-level fluctuations (Azous and Richter 1995, Richter and Azous 1995, Reinelt et al. 1998, Reinelt and Taylor 2001). Agriculture may also impact wetland hydroperiods. Euliss and Mushet (1996) compared water-level fluctuations in prairie-pothole-region wetlands surrounded by grassland and tilled agricultural land. Water-level fluctuations were greater in tilled areas, a characteristic that was attributed to increased surface runoff in the tilled landscapes.

Water Quality

Although several studies have related variations in plant- and animal-community structure to the proximity of altered landscapes, not many have assessed the effect of water-quality impacts, including nutrient enrichment, on wetlands. Bedford et al. (1999) conducted a literature review concerning the potential response of North American wetlands to nutrient enrichment. They concluded that few studies have quantified the relationship between species composition and diversity and direct measures of nutrient availability, indicating that the more common approach used was to relate these community attributes to hydrologic and chemical gradients. Some studies concerned with vegetation and nutrient relations inferred nutrient supply from plant biomass. Given these limitations, Bedford et al. (1999) indicated that existing studies confirmed the generalization that plant-community types change across broad nutrient gradients, although this broad statement does not always apply within community types. That is, results based on natural-plant-community gradients may not apply to gradients imposed within a particular plant community by human-induced nutrient enrichment.

Agriculture can directly affect the chemistry of wetlands. Nutrient enrichment of streams associated with increased erosion was the result of the conversion of bottom-land-hardwood forest to agricultural fields in the Tensas Basin of Louisiana (Childers and Gosselink 1990). Submergent plant-species richness in Great Lakes basin marshes decreased along a nutrient-enrichment and turbidity gradient primarily related to an increase in the percentage of agricultural land in the associated watersheds (Crosbie and Chow-Fraser 1999, Lougheed et al. 2001). Macrophyte-community structure in these marshes was also influenced by nutrient-enrichment and turbidity, with several taxa, including *Typha* (cattails), *Sagittaria* (arrowheads), *Nuphar variegata* (spatterdock), and *Nyphaea odorata* (white water lily), tolerating turbid and nutrient-rich waters.

Houlahan and Findlay (2004) related the extent of forest cover, cropland, building density, and road density to standing-water and surface-sediment nutrient levels in Canadian wetlands. Total Kjeldahl-nitrogen and total phosphorus concentrations in water samples were positively correlated with building density within 2,500 m (8,202 ft) and cropland within 4,000 m (13,123 ft) of the wetland edge and a negatively correlated with forest cover out to 2,250 m (7,382 ft). Sediment-phosphorous concentrations were negatively correlated with wetland size and forest cover at 4,000 m and positively correlated with the percentage of wetland within 4,000 m. Based on these results, Houlahan and Findlay

(2004) concluded that natural, regional landscapes, rather than narrow buffers, are needed to adequately conserve wetlands.

Zhu and Ehrenfeld (1999) studied nitrogen-cycling processes in Pinelands cedar swamps. Two swamps, located in the North Branch Forked River basin, were surrounded by unsewered, residential development and received stormwater discharges from paved roads. Two other sites were located in the Mount Misery Brook watershed within state-forest land. Compared to the state-forest swamps, the suburban swamps had higher pH, higher mineral-ash content, lower soil-water content, and higher total Kjeldahl nitrogen (TKN) in the soil. Nitrogen mineralization (ammonification) and nitrification were higher in the suburban swamps. Zhu and Ehrenfeld (1999) attributed the higher nitrification rates at suburban swamps to the higher pH at these sites, which may be associated with mineral ash transported in storm runoff.

In a study of Canadian wetlands, Moore et al. (1989) suggested that infertile wetlands, a characteristic shared by Pinelands wetlands, are more sensitive to human disturbances. Ehrenfeld and Schneider (1991) found that suburbanization had a significant effect on water chemistry in Atlantic white cedar swamps. Compared to cedar swamps in undisturbed watersheds, ammonia levels in surface and groundwaters of swamps located adjacent to residential development using septic systems or similar sites with direct-stormwater discharges to the wetlands were substantially higher. Elevated orthophosphate concentrations were also found in both ground and surface waters of swamps receiving stormwater runoff. Chloride and lead also increased along the disturbance gradient. Changes in water quality were more important in determining changes in community composition and structure than were changes in hydrology.

Plant communities may be more susceptible to invasion when severe disturbance is accompanied by nutrient enrichment (Burke and Grime 1996), although the evidence supporting this hypothesis is limited (Galatowitsch et al. 1999). Owen (1999) associated the spread of reed canary grass (*Phalaris arundinacea*) and cattails (*Typha angustifolia*, *T. latifolia*, and the hybrid *Typha X. glauca*) and the decline of sedges (*Carex* spp.) with wetland alterations and increasing urbanization occurring in a southern Wisconsin watershed from 1850 to 1990. The changes in vegetation were attributed to altered hydrology and possibly water-quality degradation. Experimental evidence suggests that the dominance of invasive reed canary grass over native-sedge-meadow communities in North American prairie wetlands is enhanced by nitrate-nitrogen enrichment associated with agriculture (Green and Galatowitsch 2002). Hydrologic alterations may also facilitate invasion by some nonnative plant species with plastic growth strategies that allow them to respond to variable hydrology (Galatowitsch et al. 1999). Discharge of secondarily treated wastewater to a diverse spruce-sedge-leatherleaf peatland in Michigan resulted in a shift to a nearly pure stand of cattails (Kadlec and Bevis 1990). In the Pinelands, *Typha latifolia* is generally associated with streams characterized by degraded water quality in watersheds with a high percentage of developed and upland agricultural land (Figure 1.14, Zampella and Laidig 1997, Zampella et al. 2001).

Japanese stiltgrass (*Microstegium vimineum*) is a widespread, invasive grass that was introduced to the United States from Asia in the early part of the 20th century (Barden 1987, Ehrenfeld 1999, Fairbrothers and Gray 1972, Hunt and Zaremba 1992, Snyder and Kaufman 2004). This shade-tolerant species (Winter et al. 1982, Horton and Neufeld 1998), which is associated with floodplains and disturbed areas (Barden 1987, Redman 1995, Hunt and Zaremba 1992), is frequently implicated in the displacement of native vegetation (Barden 1987, Leicht et al. 2005). Like cattail, Japanese stiltgrass is associated with degraded Pinelands streams (Figure 1.14, Zampella and Laidig 1997, Zampella et al. 2001).

Previously cited Pinelands stream-vegetation studies (Zampella and Laidig 1997, Zampella et al. 2001, 2003, 2005, 2006a) evaluated the composition of stream-channel and stream-bank vegetation in relation to water quality and land use. The composition of stream-bank vegetation, which is comprised primarily of wetland species rather than aquatic species and is frequently exposed to surface water, offers the best indication of the effect of water-quality degradation on wetland species in the Pinelands. In each major watershed, the presence of nonnative stream-bank species was associated with increasing pH, specific conductance, and land-use intensity.

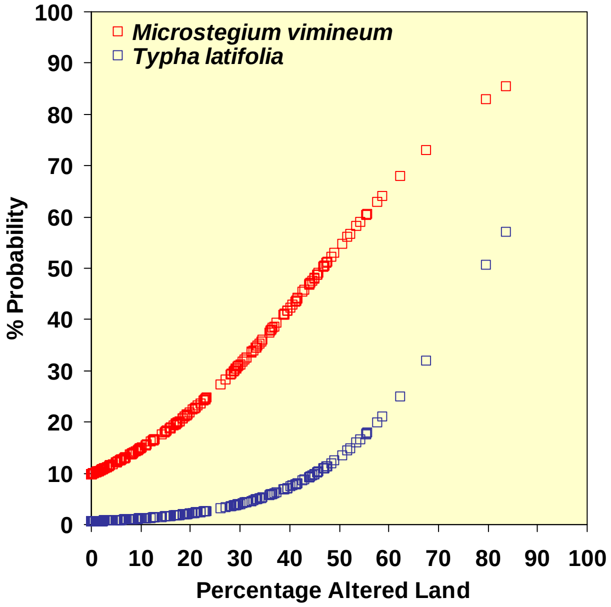

Figure 1.14. The probability of encountering *Microstegium vimineum* and *Typha latifolia* in relation to the percentage of altered land in a watershed (n = 202 stream sites). Based on data from Zampella et al. (2001, 2003, 2005, 2006a).

APPENDIX 2. MOVING-WINDOW ANALYSIS

What is a Moving-window Analysis?

Applications. A moving-window analysis, in which a “window” moves across a layer of rasterized or cell-based spatial data and characterizes the area or neighborhood surrounding each cell (Figure 2.1), has been used in ecological studies in various ways. One study employed a window to model habitat suitability for the sage-grouse (Aldridge 2003). In a study of human-caused grizzly-bear mortality, Nielsen et al. (2004) characterized terrain using a moving-window with aspect and slope layers as the inputs. Natural-vegetation-community diversity and roadless areas in a California ranch were both determined using a moving-window (Stallcup et al. 2003). In their study of great bustards (*Otis tarda*) in Spain, Osborne et al. (2001) used a moving-window analysis to identify suitable habitat patches for the species. Jones et al. (1997) used a moving-window analysis to calculate road density, forest edge, and interior habitat in watersheds in the mid-Atlantic region of the United States. Riitter et al. (2002) characterized fragmentation across the conterminous United States using square windows of varying sizes.

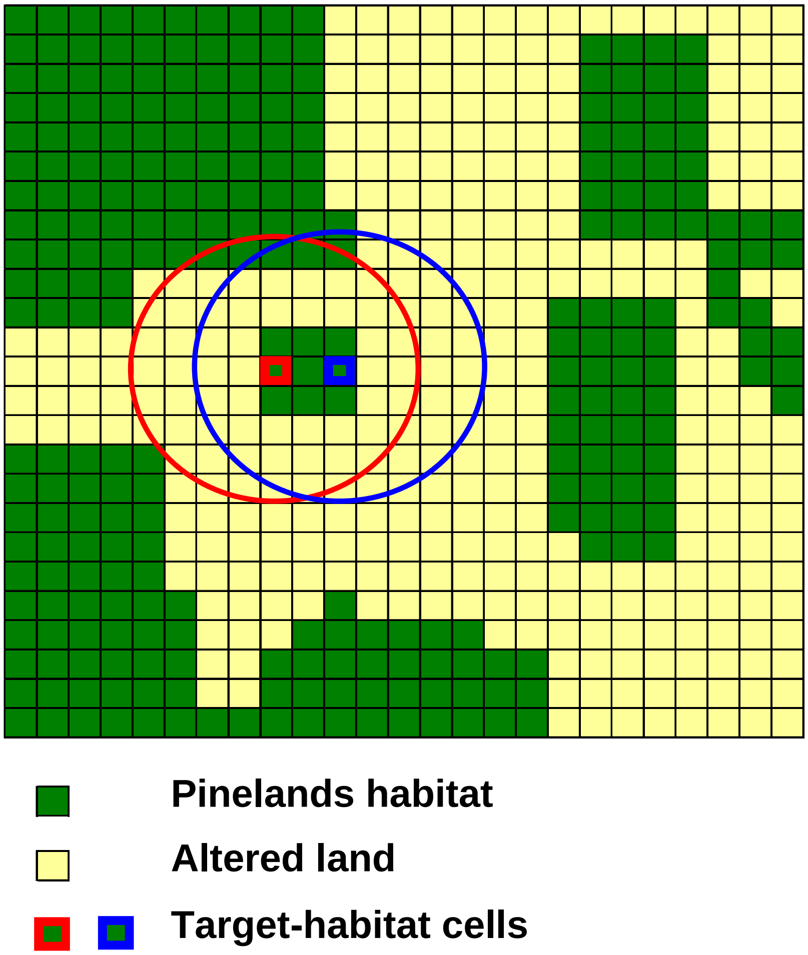

Figure 2.1. An example of a moving window. The window moves across a layer of rasterized or cell-based spatial data, performs a specified calculation on the data in the window, assigns the result of that calculation to the center cell in the window, and moves on to the next cell.

Selecting a Window Size

The importance of scale. The scale at which the Pinelands landscape was evaluated was a major consideration when selecting the size of the moving window used in the landscape-integrity assessment. When choosing an appropriate neighborhood for a moving-window, both local habitats and broader-landscape characteristics were considered. As the size of a neighborhood increases, the likelihood of adequately characterizing local habitats decreases. Conversely, too small a window minimizes the importance of surrounding landscapes. Ideally, the area of influence or neighborhood should reflect the dispersal capacity of a particular organism (Bender et al. 2003) or represent some minimum area that would ensure the viability of a particular Pinelands species, group of species, or natural community. Unfortunately, the dispersal capacity of a species or the minimum area needed to maintain even a single species or a subset of communities is generally unknown. Ultimately, the selection of a window size is a subjective decision.

Minimizing variation. Because different window sizes can produce different results (Riitters et al. 2002), it is important to select a window that reduces this variation. Preston (2002) determined optimal window sizes by quantifying the variation associated with stepwise increases in neighborhood size and selecting the neighborhood that minimized this variation. He defined no significant variation as less than a one-percent change in mean values among 40 calibration points and less than a five-percent change in value at 95% of the individual calibration points. A similar approach was used as the initial step in determining the optimal-neighborhood size for the Pinelands landscape-integrity assessment.

Three-hundred random points that fell on Pinelands habitat (upland forest, water, and wetlands, excluding managed wetlands and wetland agriculture) within the Pinelands Area were selected for analysis (Figure 2.2).[5] No point occurred within 3,000 m (9,842 ft) of another. Fifteen windows of varying sizes, represented by concentric circles with radii ranging from 100 to 1,500 m (328 to 4,921 ft), were created around each point. For each random-sample point, the percentage of Pinelands habitat within each of the concentric circles or windows was determined and the change in habitat extent that resulted with each 100-m increase in window radius was calculated. Results of the analysis indicated that the average change in the percentage of Pinelands habitat for the 300 windows decreased to about one percent between a window radius of 900 m and 1,000 m (2,953 ft and 3,281 ft, Figure 2.3). The slope of the average-percentage change also began to level off between 1,000 m and 1,100 m (3,281 ft and 3,609 ft). Additionally, a Spearman rank correlation matrix relating the percentage of Pinelands habitat in each group of window-size classes to all other window-size classes was constructed and the median correlation coefficient (r) of each window-size class was calculated. Overall, the 1,000-m window displayed the strongest relationship (median $r = 0.97$, $p<0.001$) with the other window-size classes (Figure 2.4). Based on these results, the 1,000-m-radius (314-ha or 776-acre) window was selected for further analysis.

Relationship of 1,000-m-radius window to landscape metrics. The extent of Pinelands habitat in the 1,000-m-radius windows was related to several other important landscape metrics (Figure 2.5). For the 300 1,000-m-radius windows, the relationship between the proportion of Pinelands habitat and edge was bell-shaped, which is consistent with the pattern described by Fahrig (2003). Edge, which was measured as the length of adjacency between Pinelands habitat and non-habitat, increased with the proportion of Pinelands habitat up to about 50%. In these cases, Pinelands habitat patches are embedded in a developed/agricultural matrix. Edge decreased as Pinelands habitat became the dominant cover type. The relationship between the proportion of

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[5] The Hawth's Analysis Tools Extension Version 3.27 Random Point Generator in ArcGIS version 9.2 (ESRI 1999-2006) and 2002 land-use/land-cover vector data (NJDEP 2007) were used to select the random-habitat points.

Pinelands habitat and mean Pinelands-habitat-patch size was positive exponential, whereas Fahrig (2003) described this association as linear. The relationship was linear when the log of the mean patch size was considered. The number of patches decreased as the proportion of Pinelands habitat increased, whereas Fahrig (2003) described this relationship as a skewed, bell-shaped curve, a pattern that might be expected in landscapes more highly altered than those found in the Pinelands.

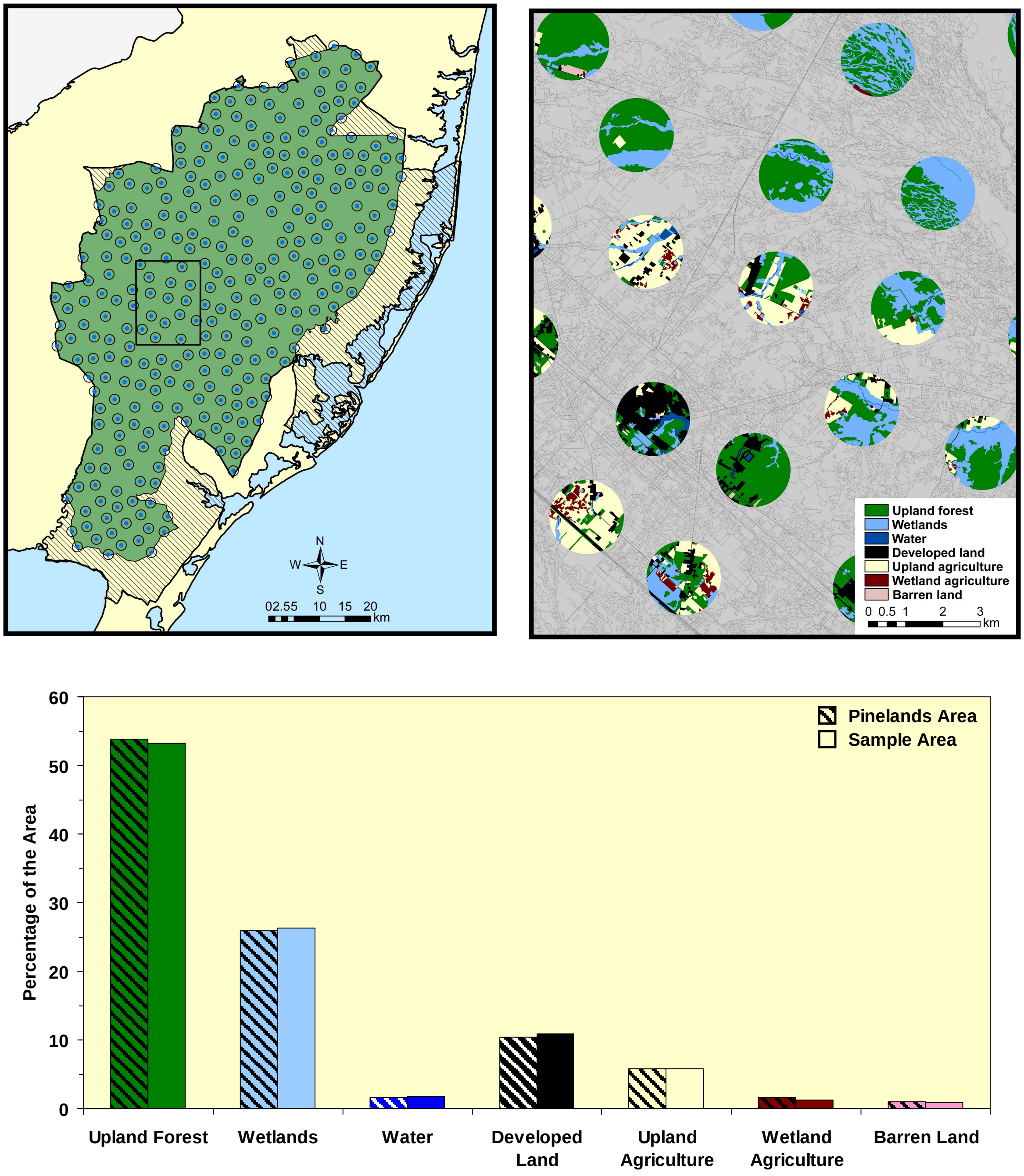

Figure 2.2. A random sample of 300 points that fell on Pinelands habitat within the Pinelands Area and 1,000-m-radius windows surrounding the points. The sample encompassed 94,248 ha (232,892 acres) and captured the full diversity of Pinelands land uses.

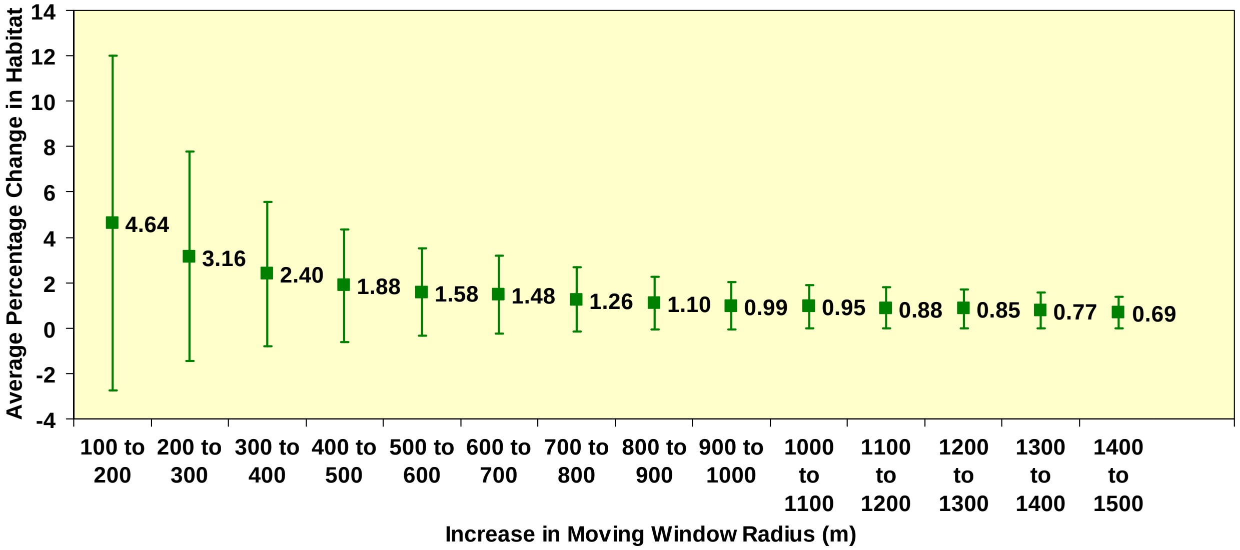

Figure 2.3. The average change in the percentage of Pinelands habitat as the radius of 300 windows increases from 100 to 1,500 m.

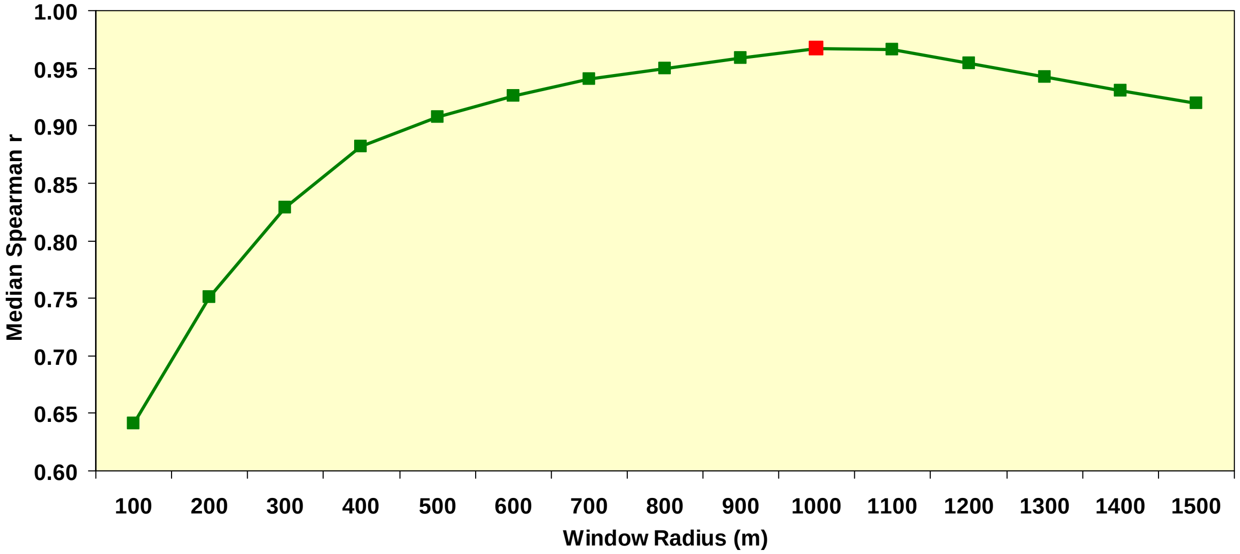

Figure 2.4. Median Spearman rank correlation coefficient (r) of each window-size class (100-m radius, 200-m radius, etc.) based on a correlation matrix relating the percentage of Pinelands habitat in each group of window-size classes to all other window-size classes (n = 300).

Biological basis. Several studies suggest that a 1,000-m-radius window may be an appropriate scale for a landscape analysis relating land use to ecological integrity. Land use within 1,000 m has been associated with variations in amphibian-species richness (Knutson et al. 1999, Lehtinen et al. 1999, Herrmann et al. 2005), the probability of occurrence of breeding-bird species (Darr et al. 1998), the density and percentage of wintering-alien birds (Smith 2007), and bird and wetland-plant diversity (Mensing et al. 1998). Rodewald and Yahner (2001) considered a 1,000-m radius spatial scale large enough to be perceived as a landscape by most forest birds.

Neighborhood characteristics can also be evaluated in relation to the needs of an umbrella species or a suite of focal species, a multi-species concept that is an extension of the umbrella-species idea, with the assumption that such species encompass the habitat requirements of other co-occurring species (Lambeck 1997, Turner et al. 2001, Roberge and Angelstam 2004). In the Pinelands, wide-ranging species with different habitat needs and the greatest area requirements, such as the barred owl (*Strix varia*), timber rattlesnake (*Crotalus horridus*), and northern pine snake (*Pituophis m. melanoleucus*), represent a suite of umbrella species that occupy a variety of habitats used by many other Pinelands species.

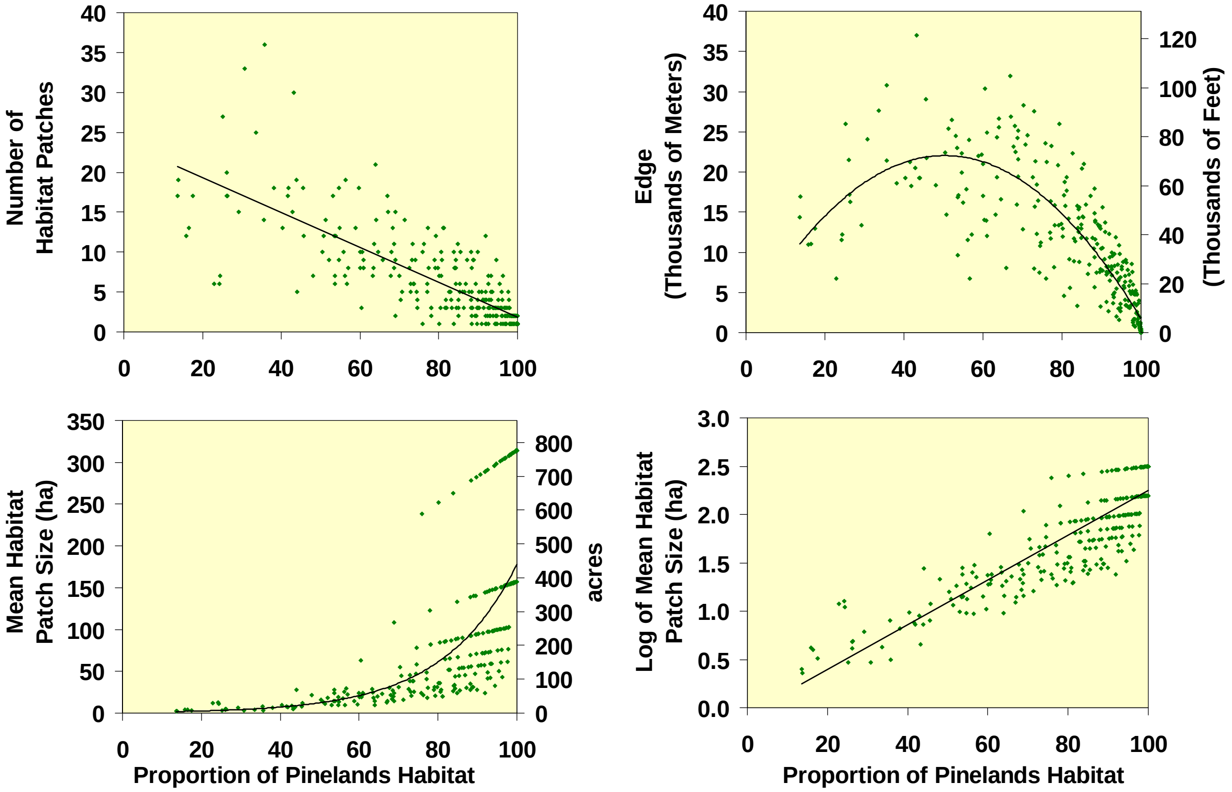

Figure 2.5. The relationship between the proportion of Pinelands habitat and edge, number of habitat patches, and mean patch size.

In a species-based approach, home-range data can be used to delineate a buffer area around an individual sighting for a species, but this approach has several limitations. First, the approach requires sighting data. Factors such as the age of data points in a rapidly changing landscape, uneven sampling distribution across the landscape, and coverage that is representative of only a small portion of species, limit the usefulness of such data even when they are available. Secondly, the size and geometry of a home range varies among individuals and populations of the same species and is influenced by landscape characteristics. A circular buffer may not reflect the natural geometry of the home range, and using a sighting as the center of a buffer provides only one of many possible configurations. For example, the home range of the barred owl is usually an irregularly shaped patch that follows natural- and human-altered-landscape features and varies between regions (Elody and Sloan 1985, Hegdal and Colvin 1988, Nicholls and Warner 1972, Mazur et al. 1998, Nicholls and Fuller 1987). Home-range size and geometry can also vary between males and females of the same species as demonstrated in studies of two widely separated Pinelands rattlesnake populations (Reinert and Zappalorti 1988, Laidig and Golden

2004). Furthermore, a home-range-based buffer may offer some protection to an individual for a period of time, but may not ensure the long-term maintenance of a population. Once the individual dies, the value of the home-range buffer becomes questionable. How to use individual sightings and home-range buffers to evaluate habitat outside the buffer presents another problem since, by definition, the home range, not the habitat extending beyond it, is the area used by the individual. Fewer limitations are associated with the use of buffers around stationary sites such as a nesting area or an amphibian-breeding pond and core-terrestrial habitat, although the issue of how to apply this information at the landscape level remains unresolved.

A moving-window analysis provides an opportunity to use home-range or breeding-pond information to evaluate landscape integrity independent of actual-sighting data by determining the total amount of habitat surrounding all possible sites where an umbrella species, such as the barred owl, northern pine snake, timber rattlesnake, or a hypothetical wide-ranging species that uses a variety of upland and wetland habitats, could possibly be found. The 314-ha area (776 acres) within a 1,000-m-radius window encompasses circular buffers based on reported home ranges for the pine snake and timber rattlesnake and some of those home ranges given for the barred owl (Figure 2.6).

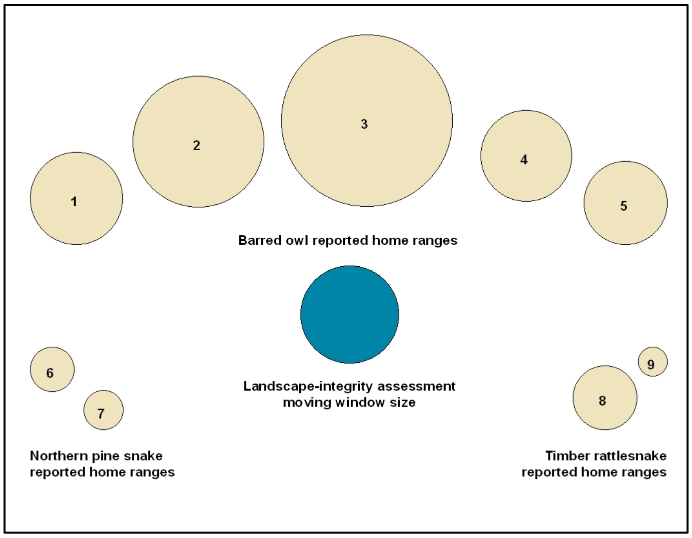

Figure 2.6. Comparison of a 1,000-m window and published home ranges for the barred owl, northern pine snake, and timber rattlesnake. Sources: 1) Elody and Sloan (1985), Michigan; 2) Hegdal and Colvin (1988), Virginia; 3) Mazur et al. (1998), Saskatchewan, Canada; 4) Nicholls and Fuller (1987), Minnesota; 5) Nicholls and Warner (1972), Minnesota; 6) Gerald et al. (2006), Tennessee; 7) Zappalorti et al. (2001), New Jersey; 8) Laidig and Golden (2004), New Jersey; 9) Reinert and Zappalorti (1988), New Jersey.

Accounting for proximity. Proximity of land-use activities can affect some species (Appendix 1). One way to incorporate proximity into a moving-window analysis is to use an inverse-distance-weighted approach, where the inverse distance from a target cell to every other Pinelands-habitat cell or non-habitat cell is determined. This approach gives greater weight to cells near the target cell while the influence of more distant cells decreases exponentially. A second approach weights each Pinelands-habitat cell using linear distance, where the cell closest to the target cell is assigned a weight of 100 and the cell furthest from the target cell is assigned a weight of one. In this case, the importance of a cell decreases linearly from the target cell at the center of the window. Both approaches were used to determine distance-weighted Pinelands-habitat values within 1,000-m-radius windows surrounding the same 300 random points used in the previous analyses. However, unlike the previous analyses, the distance-weighted analyses were based on rasterized data.

Using Spearman rank correlation, the distance-weighted results were compared to those based simply on the extent of unweighted habitat within a target cell's neighborhood. The results obtained using unweighted-habitat values were highly correlated with those using the inverse-distance (r = 0.99, $p < 0.001$) and linear-distance approaches (r = 0.96, $p < 0.001$). Unweighted-habitat values were used in the landscape-integrity assessment because this approach was the simplest of the three methods, both conceptually and computationally, and yielded results almost identical to the two distance-weighted approaches.

Summary

A 1,000-m-radius moving window was selected for use in the landscape-integrity assessment for several reasons. Among these are the results of several ecological studies that suggested that a 1,000-m-radius window might be an appropriate scale for a landscape analysis relating land use to ecological integrity. An analysis of variable-window sizes indicated that the use of a 1,000 m-radius window minimized the effect of changing the scale at which the landscape is evaluated. Additionally, the extent of Pinelands habitat in the 1,000-m-radius windows was related to several other important landscape metrics, including edge, the number of patches, and mean patch size. Finally, the 314-ha (776 acre) area within a 1,000-m-radius window encompassed circular buffers based on some reported home ranges for wide-ranging species such as timber rattlesnake, northern pine snake, and barred owl. These three species represent umbrella species that may encompass the habitat requirements of other co-occurring species. Because proximity of land-use activities can affect some species, the effect of distance of adjacent habitat to target cells was also analyzed. The results indicated that the total amount of habitat within a neighborhood was related to distance-weighted habitat values and provided a simpler, straightforward measure of adjacent habitat.

APPENDIX 3. EVALUATING WETLAND-DRAINAGE-UNIT LAND-USE PATTERNS

To determine if the proximity of altered land influences the relative comparison of wetland-drainage units, land-use patterns in a random sample of 500 units from the pool of 12,516 wetland-drainage units were analyzed. The percentage of altered land (developed land and upland agriculture) in each of a series of cumulative upland buffers surrounding each wetland unit was determined. The buffers were increased in 50-m (164-ft) increments up to a maximum width of 500 m (1,640 ft).

Uplands were absent in 35 of the wetland-drainage units and the size of the largest upland buffer varied among the remaining 465 units. Analysis of the 465 wetland-drainage units revealed that the change in the percentage of altered land varied little as the upland-buffer width increased, which suggests that altered land was distributed fairly evenly in the upland portion of the units (Table 3.1). Additionally, the correlations among the percentages of altered land in entire wetland-drainage units, the upland portion of wetland-drainage units, and individual cumulative-buffers were strong and highly significant (Table 3.2).

Table 3.1. The mean percentage of altered land (developed land and upland agriculture) in a random sample of 465 wetland-drainage units. Mean altered-land values are given for entire wetland-drainage units, the upland portion of wetland-drainage units, and individual cumulative-upland buffers. Uplands were not present in 35 units included in the initial sample of 500 wetland-drainage units and the number of buffers varied among units. Conversion: 1 m = 3.28 ft.

		Percentage of altered land in entire drainage units, upland areas, and cumulative buffers											
				Cumulative buffers (m)									
Number of buffers	Number of drainage units	Entire unit	Upland area	50	100	150	200	250	300	350	400	450	500
1	31	5.2	36.2	36.2									
2	64	8.2	23.6	23.3	23.6								
3	33	7.6	15.3	16.0	15.5	15.3							
4	39	7.9	10.4	10.5	10.4	10.3	10.4						
5	39	9.6	17.3	17.4	17.4	17.3	17.3	17.3					
6	30	13.2	16.8	17.9	17.4	17.2	16.9	16.8	16.8				
7	42	13.6	17.3	15.4	16.2	16.8	17.1	17.1	17.2	17.3			
8	26	14.5	17.9	15.5	15.1	15.5	16.3	17.3	17.8	17.9	17.9		
9	14	8.1	10.3	13.9	11.8	10.7	10.2	10.8	11.0	10.6	10.2	10.3	
10	147	17.7	20.1	19.0	18.3	18.7	19.1	19.3	19.4	19.5	19.6	19.7	19.8
Overall mean		12.2	19.4	18.9	17.4	16.5	16.8	17.9	18.1	18.4	18.6	18.9	19.8
SD		21.6	30.9	32.1	29.9	28.2	28.2	28.3	28.1	27.9	27.1	26.8	27.4
Total n		465	465	465	434	370	337	298	259	229	187	161	147

To further explore the effect of land-use proximity on the altered-land value assigned to a wetland-drainage unit, the percentage of altered land in each individual buffer ring (i.e., 0-50 m, 50-100 m, etc.) in all 465 random samples was weighted by multiplying this value by: 1) the inverse of its position relative to the wetland boundary, where the first buffer (50 m or 164 ft) was assigned a weight equal to 1 and the tenth buffer (500 m or 1,640 ft) was assigned a weight of 1/10 or 0.1, and 2) the buffer-ring percentage contribution to the entire upland area. The individual buffer-ring-weighted values were summed to produce a weighted altered-land value for each wetland-drainage unit. Employing the same approach, weighted altered-land-use values were also produced using the percentage contribution of each ring to the entire drainage-unit area and to the largest cumulative-buffer area, which in some cases was less than the entire upland portion of the wetland-drainage unit. The weighted percentages obtained using all three methods were significantly correlated with the percentage of altered land in the largest-cumulative buffer, the upland portion of the wetland-drainage unit, and the entire wetland-drainage unit (Table 3.3).

This exercise does not demonstrate that proximity has no effect on wetland-drainage integrity. However, it does indicate that existing land-use patterns in the Pinelands are such that the relationship between wetland-drainage units is similar regardless of whether near, far, or total altered land is considered, and that there is little difference when comparing wetland-drainage units using the percentage of altered land in a particular upland buffer rather than the percentage of altered land in either all the associated uplands or the entire wetland-drainage unit. Because including only uplands would exaggerate the possible influence of a high percentage of altered land in wetland-drainage units with a low percentage of uplands, the percentage of altered land in the entire wetland-drainage unit was selected as the primary measure of wetland-drainage integrity.

Table 3.2. Spearman rank correlations (r) between the percentage of altered land (developed land and upland agriculture) in entire wetland-drainage units, the upland portion of wetland-drainage units, and individual, cumulative upland buffers in a random sample of 465 wetland-drainage units. Sample size varies between comparisons. All correlations are significant at $p < 0.001$. Conversion: 1 m = 3.28 ft.

Cumulative upland buffer (m)	Entire unit	Entire upland area	Cumulative upland buffers (m)									
			50	100	150	200	250	300	350	400	450	500
50	0.86	0.91										
100	0.91	0.94	0.98									
150	0.92	0.94	0.96	0.99								
200	0.93	0.94	0.94	0.98	1.00							
250	0.93	0.95	0.93	0.97	0.99	1.00						
300	0.94	0.95	0.91	0.95	0.98	0.99	1.00					
350	0.95	0.96	0.89	0.94	0.96	0.98	0.98	0.99				
400	0.95	0.96	0.86	0.92	0.95	0.97	0.97	0.98	1.00			
450	0.96	0.96	0.85	0.90	0.94	0.95	0.96	0.97	0.99	1.00		
500	0.97	0.97	0.84	0.90	0.92	0.94	0.95	0.96	0.98	0.99	1.00	
Upland area	0.96	-	0.91	0.94	0.94	0.94	0.95	0.95	0.96	0.96	0.96	0.97
Entire Unit	-	0.96	0.86	0.91	0.92	0.93	0.93	0.94	0.95	0.95	0.96	0.97

Table 3.3. Spearman rank correlations (r) relating distance-weighted land-use values to the percentage of altered land within entire wetland-drainage units, the upland portion of wetland-drainage units, and the largest cumulative buffer. The width of the largest cumulative buffer ranged from 50 m to 500 m (164 ft to 1,640 ft). All correlations are significant at $p < 0.001$.

Percentage of altered land weighted by proximity and the:	Percentage of altered land in the:		
	Largest buffer	Uplands	Wetland-drainage unit
Largest cumulative-buffer area	0.99	0.98	0.93
Upland area	0.99	0.98	0.93
Wetland-drainage unit area	0.97	0.96	0.97

APPENDIX 4. LANDSCAPE-, AQUATIC-, AND WETLAND-DRAINAGE-INTEGRITY ASSESSMENTS

Integrity Assessments

As explained in the main body of this report, Pinelands ecological integrity is a composite of the results of the landscape-, aquatic-, and wetland-drainage-integrity assessments. Landscape integrity reflects the amount of Pinelands habitat, composed of upland forest, wetlands, and water, relative to the extent of non-habitat, represented by developed land, upland agriculture, wetland agriculture, managed wetlands, barren land, and roads. The approach used to determine the extent of Pinelands habitat in the landscape-integrity assessment also provides a means of characterizing the quality of Pinelands habitat with respect to both habitat-patch size and number and the edge or boundary between Pinelands habitat and land uses that could affect the integrity of the habitat.

The aquatic-integrity assessment was based on the assumption that integrity increases as the percentage of developed land and upland agriculture in a watershed decreases. The aquatic integrity of each drainage unit and associated stream segment was based on the percentage of the upstream watershed that was neither developed land nor upland agriculture.

Wetland-drainage integrity was represented by the percentage of a wetland-drainage unit that was neither developed land nor upland agriculture, with the assumption that this characteristic affects both the quantity and quality of groundwater flowing to the wetlands and the plants and animals that depend on these waters. Where the landscape assessment evaluated the amount of habitat surrounding a particular area and the aquatic assessment characterized the extent of land uses within a watershed that could influence water quality and the composition of aquatic communities, the wetland-drainage assessment was concerned with the more local effect of adjacent land use on the hydrologic and water-quality integrity of wetlands.

In each assessment, the 90.1-100% class represents the highest level of integrity, whereas the ≤10% class represents the lowest level of integrity. The results of the 2002 assessments are displayed spatially in Figures 4.1-4.3. Fifty-percent of the Pinelands Area fell within the highest landscape-integrity class (Figure 4.4). The highest aquatic- and wetland-drainage-integrity class was assigned to 49% and 62% of the Pinelands Area, respectively (Figure 4.4). The percentage values or scores given for both the aquatic-integrity and wetland-drainage-integrity assessments include both habitat and non-habitat because they were based on drainage areas, whereas the landscape-integrity scores were based on Pinelands habitat only. The percentage of the Pinelands Area displaying integrity scores in the five lowest integrity classes ranged from five percent for landscape-integrity to 11% for wetland-drainage integrity.

The same weighted-average method used in the overall ecological-integrity assessment was used to rank each Pinelands management area (Table 4.1) in the landscape, aquatic, and wetland-drainage assessments. For each of the nine management areas, the percentage of Pinelands habitat, drainage-unit, and wetland-drainage-unit area in each integrity class was multiplied by the upper range of the class (e.g., the weight for the 10.1-20% class equaled 20), the weighted percentages were summed and divided by 100, and the resulting management-area weighted-scores were ranked. In each assessment, Agricultural Production Areas, Pinelands Towns, and Regional Growth Areas displayed the lowest overall integrity, whereas the Preservation Area District, Forest Areas, and Special Agricultural Production Areas displayed the highest integrity (Figures 4.5-4.7).

The Preservation Area District, Forest Areas, and Special Agricultural Production Areas, which encompass nearly two-thirds of the Pinelands Area, accounted for 92% of the area in the 90.1-100% landscape-integrity class and 65% of the area in the 80.1-90% class (Figure 4.8). Five percent of the combined land area in these three management areas was classified as non-habitat. In contrast, Agricultural Production Areas, Regional Growth Areas, and Pinelands Towns accounted for 70% of the area in the five lowest landscape-integrity classes. Together, these three management areas encompass 18% of the Pinelands Area, but accounted for 50% of the region's non-habitat. About one-half (52%) of the combined land area in these management areas was non-habitat.

Eighty-nine percent of the area within the highest aquatic-integrity class was found within the Preservation Area District, Forest Areas, and Special Agricultural Production Areas (Figure 4.9). These three management areas also accounted for nearly two-thirds of the area in the 80.1-90% class. At the opposite end of the management-area gradient, Pinelands Towns, Agricultural Production Areas, and Regional Growth Areas included 61% of the area in the five lowest aquatic-integrity classes.

Preservation Area District, Forest Areas, and Special Agricultural Production Areas encompassed 86% of the area included in the 90.1-100% wetland-drainage-integrity class and 48% of the area in the 80.1-90% class (Figure 4.10). At the opposite end of the management-area gradient, Agricultural Production Areas, Pinelands Towns, and Regional Growth Areas, accounted for 66% of the area included in the five lowest wetland-drainage-integrity classes.

The landscape, aquatic, and wetland-drainage integrity of Regional Growth Areas, Pinelands Towns, and Pinelands Villages (i.e., development areas) in each municipality was ranked with the same weighted-average method used to rank the nine management areas. Similar patterns were found in each assessment (Figure 4.11). Development areas in municipalities such as Buena Township, Wrightstown, and Hammonton displayed some of the lowest integrity in all three assessments. At the opposite end of the integrity gradients, development areas in municipalities such as Little Egg Harbor Township, Woodland Township, and Lacey Township displayed the highest level of landscape integrity. Development areas in municipalities such as South Toms River and Lakehurst displayed higher aquatic integrity compared to landscape integrity because the upper parts of their watersheds were less developed. In contrast, development areas in Egg Harbor City, which is upstream from extensive Pinelands habitat, displayed higher landscape integrity compared to aquatic integrity.

The results obtained by ranking the landscape-, aquatic, and wetland-drainage integrity of Rural Development Areas in each municipality revealed patterns similar to those of the ecological-integrity assessment (Figure 4.12). Municipalities such as Medford Lakes, Berlin Township, and Plumsted Township displayed relatively low landscape, aquatic, and wetland-drainage integrity, whereas Upper Township, Maurice River Township, and Bass River Township displayed a high level of integrity.

Table 4.1. Pinelands management areas. The area and the percentage of the Pinelands Area (%) that it represents are given for each management area.

Management area	Code	Acres	Hectares	%
Regional Growth Area	RGA	76,472	30,960	8.2
Pinelands Town	PT	21,758	8,809	2.3
Pinelands Village	PV	25,907	10,489	2.8
Rural Development Area	RDA	113,181	45,822	12.1
Federal and Military Installation Area	FMI	47,550	19,251	5.1
Agricultural Production Area	APA	68,160	27,595	7.3
Special Agricultural Production Area	SAP	37,582	15,215	4.0
Forest Area	FA	252,950	102,409	27.0
Preservation Area District	PrA	294,612	119,276	31.4
Pinelands Area	PA	938,173	379,827	100.0

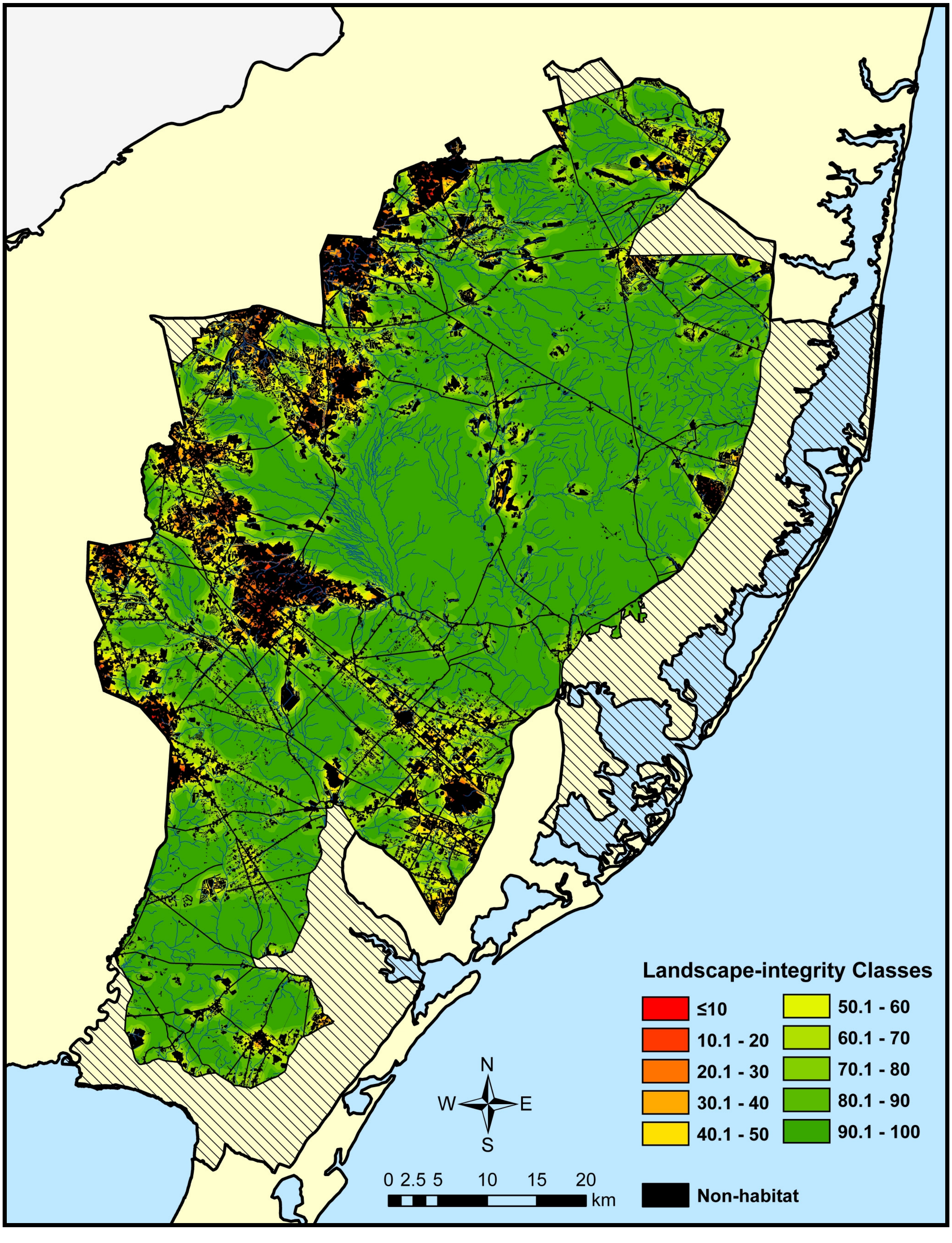

Figure 4.1. Landscape integrity based on 2002 land-use/land-cover data. The 90.1-100% class represents the highest level of landscape integrity. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

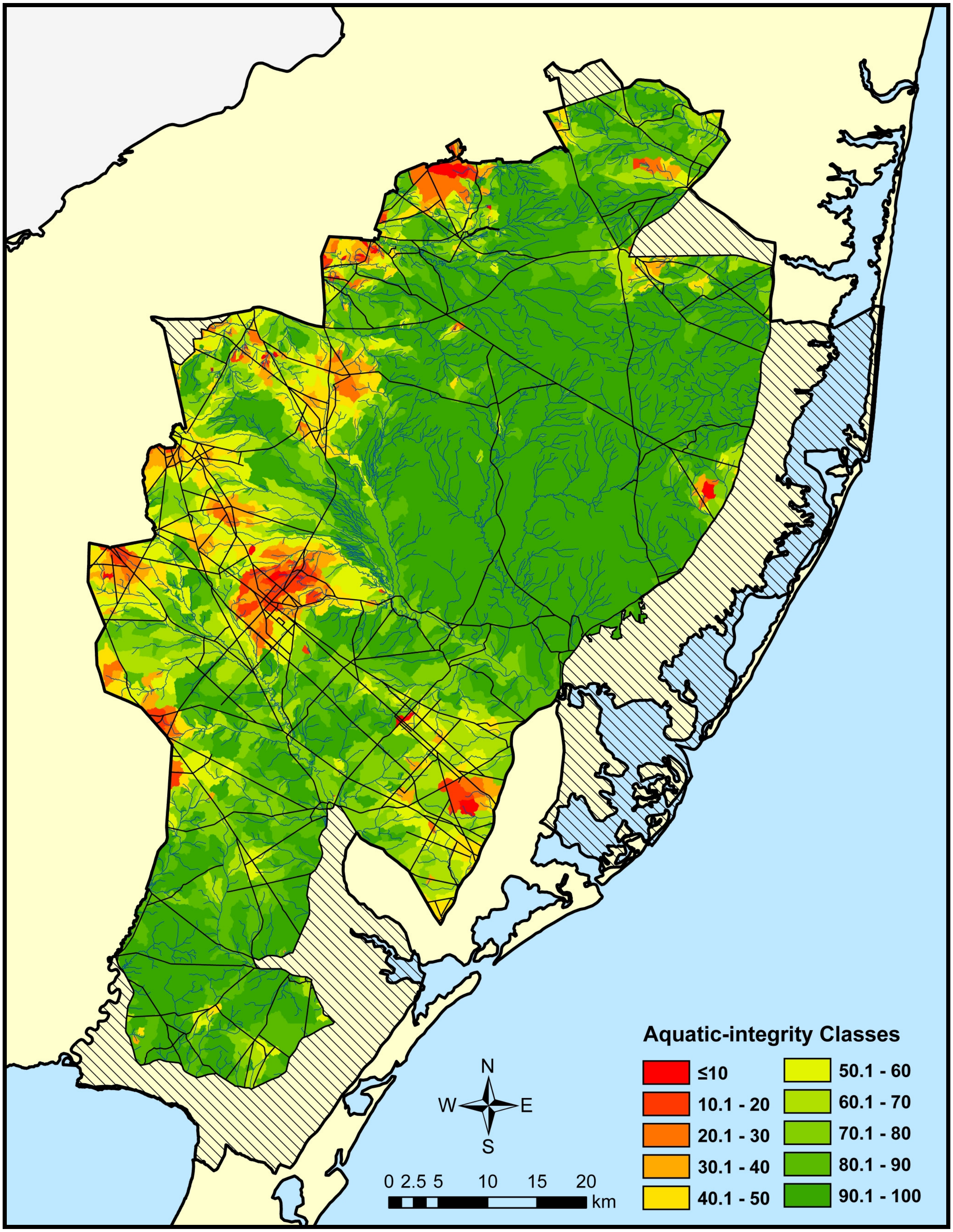

Figure 4.2. Aquatic integrity based on 2002 land-use/land-cover data. The 90.1-100% class represents the highest level of aquatic integrity. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

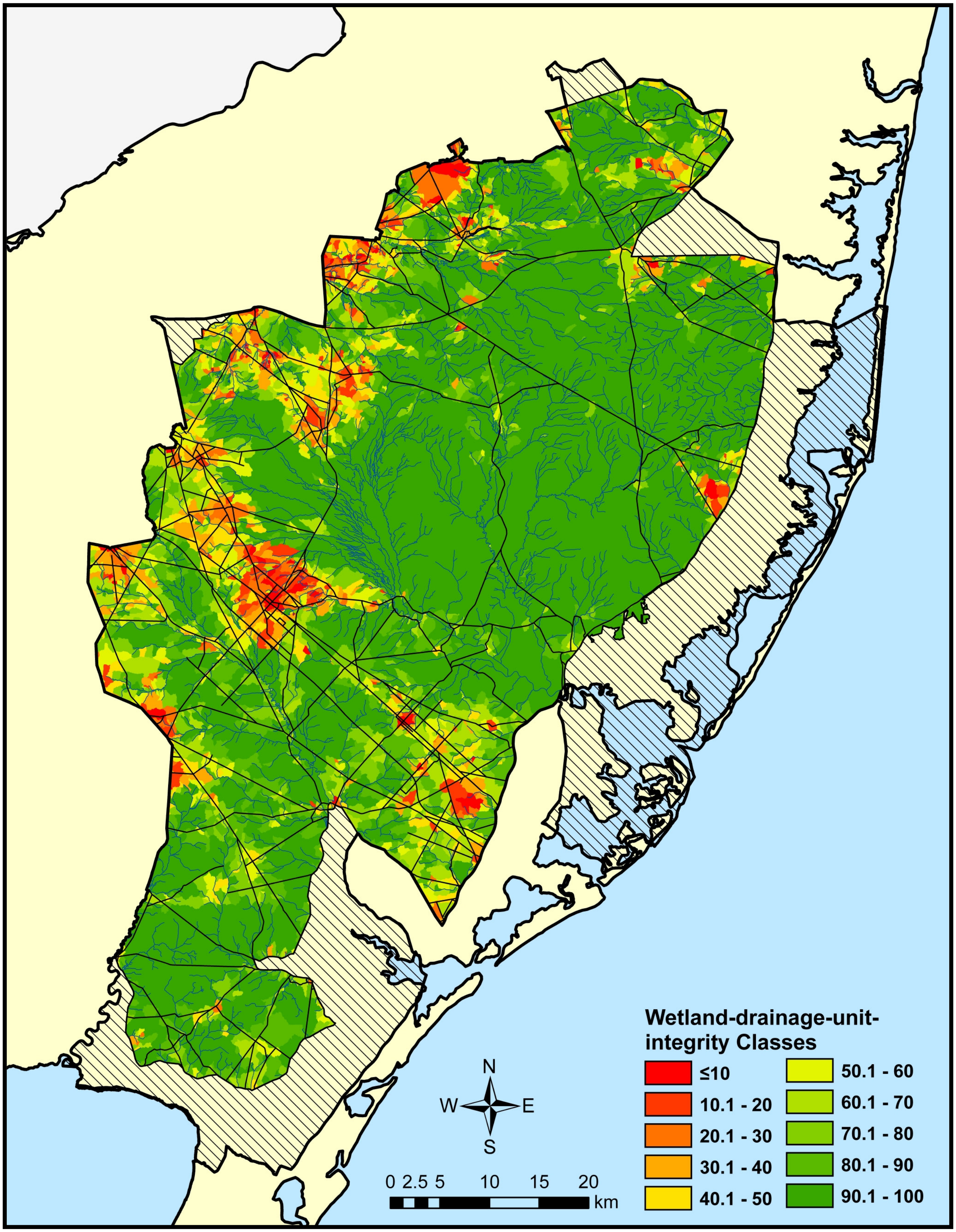

Figure 4.3. Wetland-drainage integrity based on 2002 land-use/land-cover data. The 90.1-100% class represents the highest level of wetland-drainage integrity. The hatched area represents the portion of the Pinelands National Reserve outside the Pinelands Area.

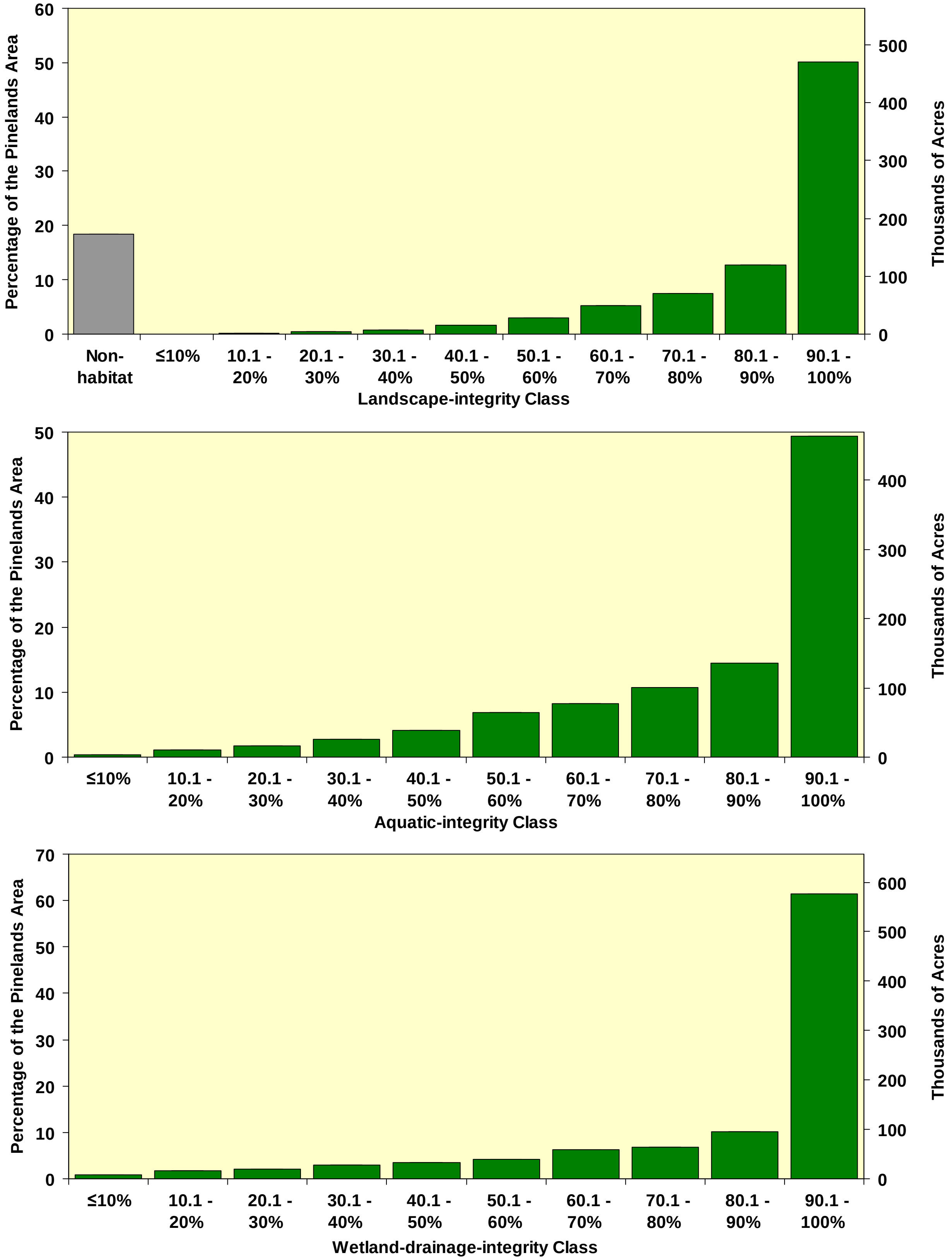

Figure 4.4. The percentage of the Pinelands Area within each landscape-, aquatic-, and wetland-drainage-integrity class. The values or scores given for both aquatic integrity and wetland-drainage integrity were assigned to entire drainage units that include both habitat and non-habitat. The landscape-integrity scores were assigned to Pinelands habitat only.

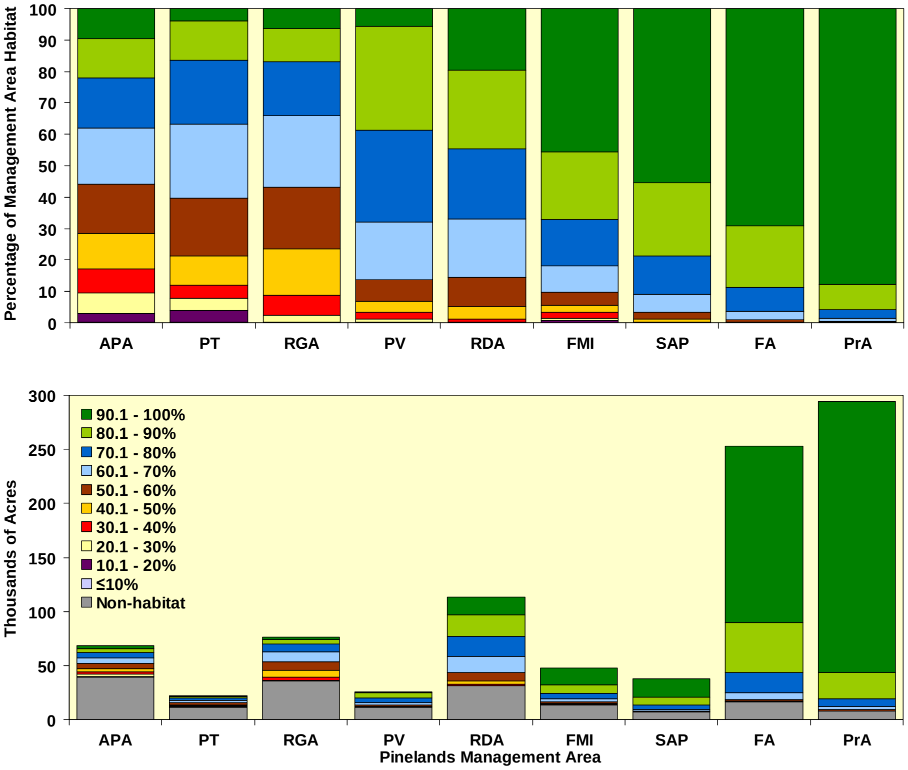

Figure 4.5. Landscape-integrity-class composition of Pinelands habitat and the acres of Pinelands habitat and non-habitat in each class by Pinelands management area. Refer to Table 4.1 for Pinelands management-area codes.

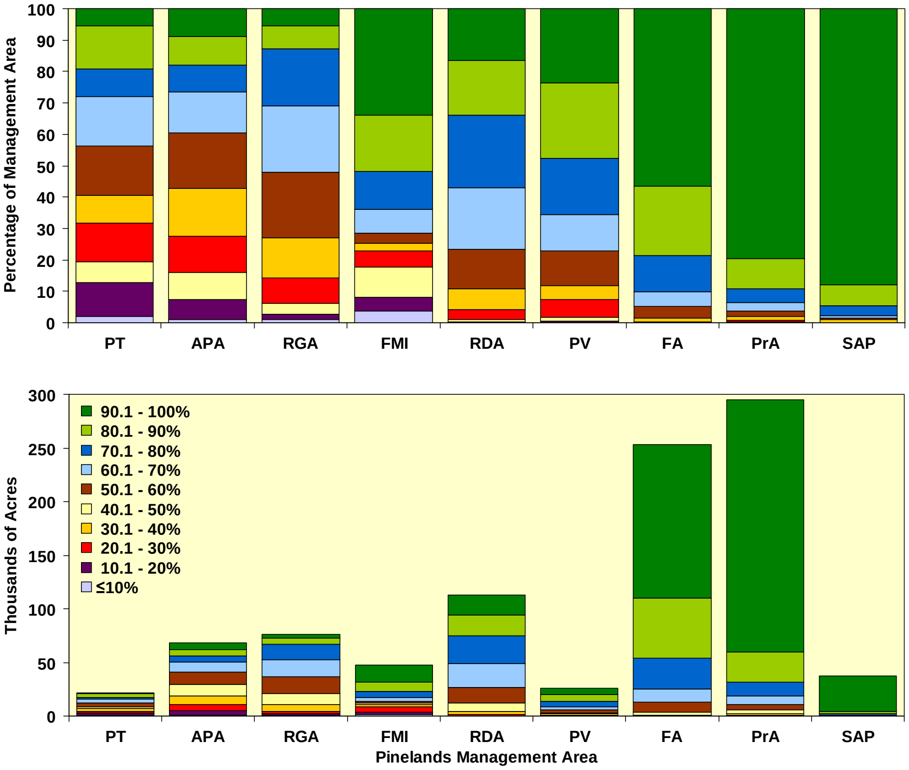

Figure 4.6. Aquatic-integrity-class composition of the Pinelands Area by Pinelands management area. The integrity values or scores were assigned to entire drainage units that include both habitat and non-habitat. Refer to Table 4.1 for Pinelands management-area codes.

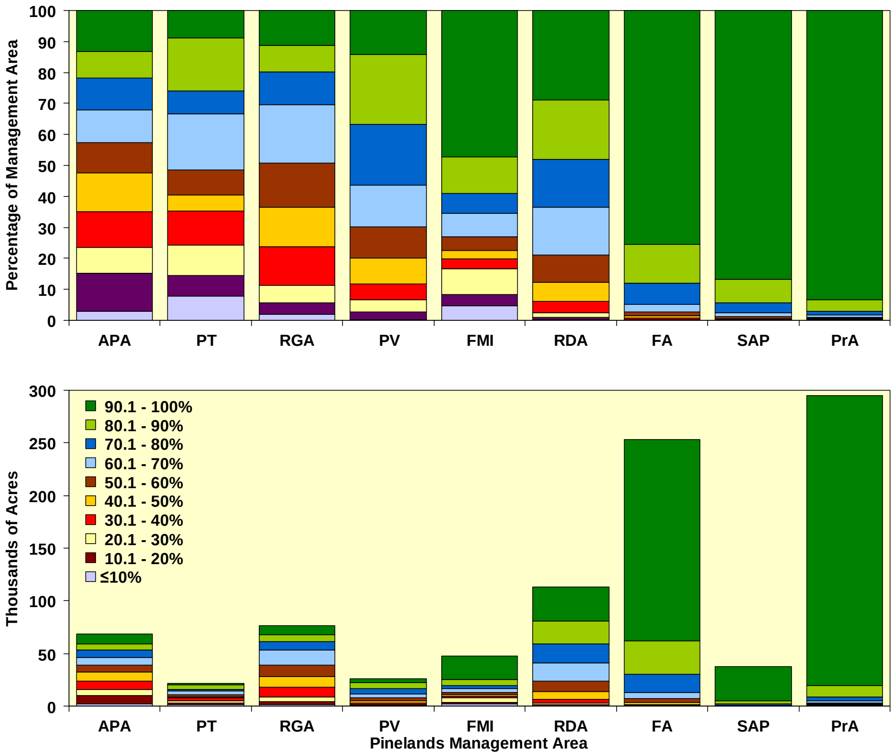

Figure 4.7. Wetland-drainage-integrity-class composition of the Pinelands Area by Pinelands Management Area. The integrity values or scores were assigned to entire wetland-drainage units that include both habitat and non-habitat. Refer to Table 4.1 for Pinelands management-area codes.

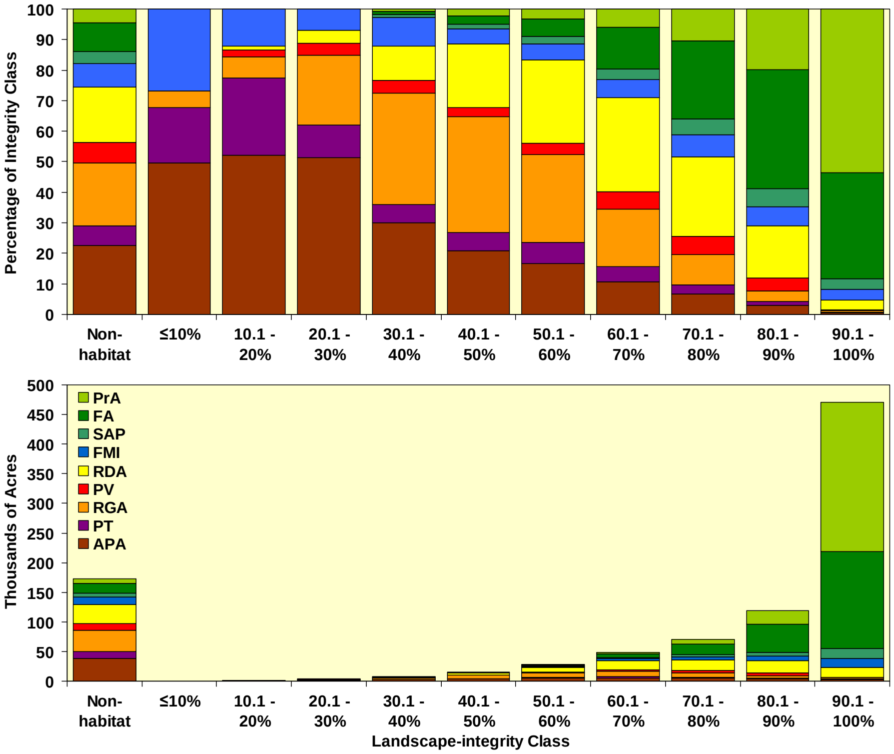

Figure 4.8. Pinelands management-area composition of landscape-integrity classes and the acres of Pinelands habitat in each class. Refer to Table 4.1 for Pinelands management-area codes.

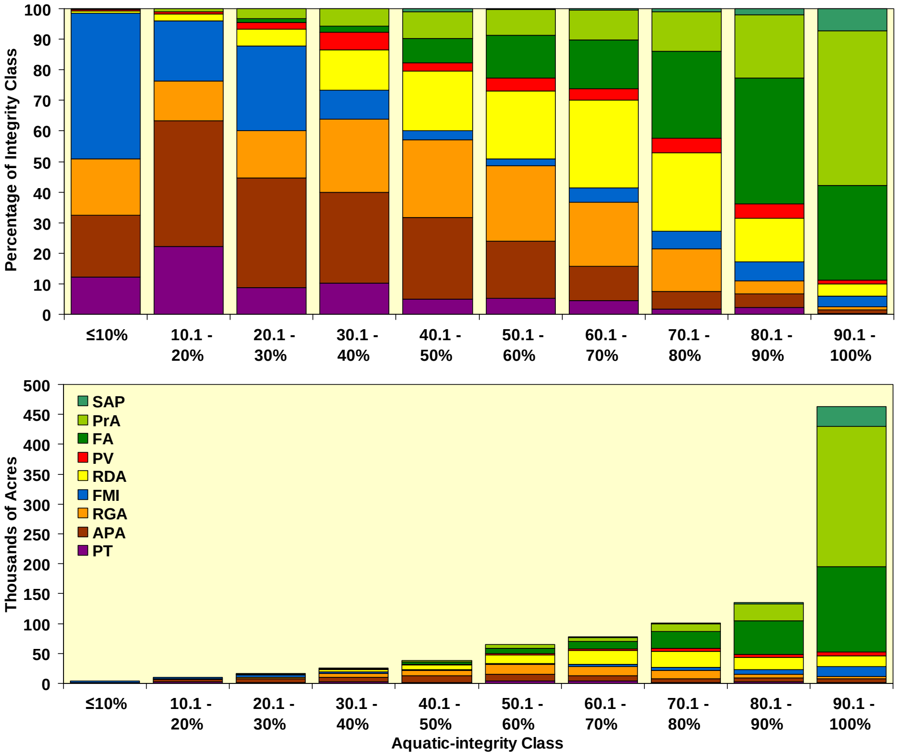

Figure 4.9. Pinelands management-area composition of aquatic-integrity classes and the acres of drainage area in each class. The integrity values or scores were assigned to entire drainage units that include both habitat and non-habitat. Refer to Table 4.1 for Pinelands management-area codes.

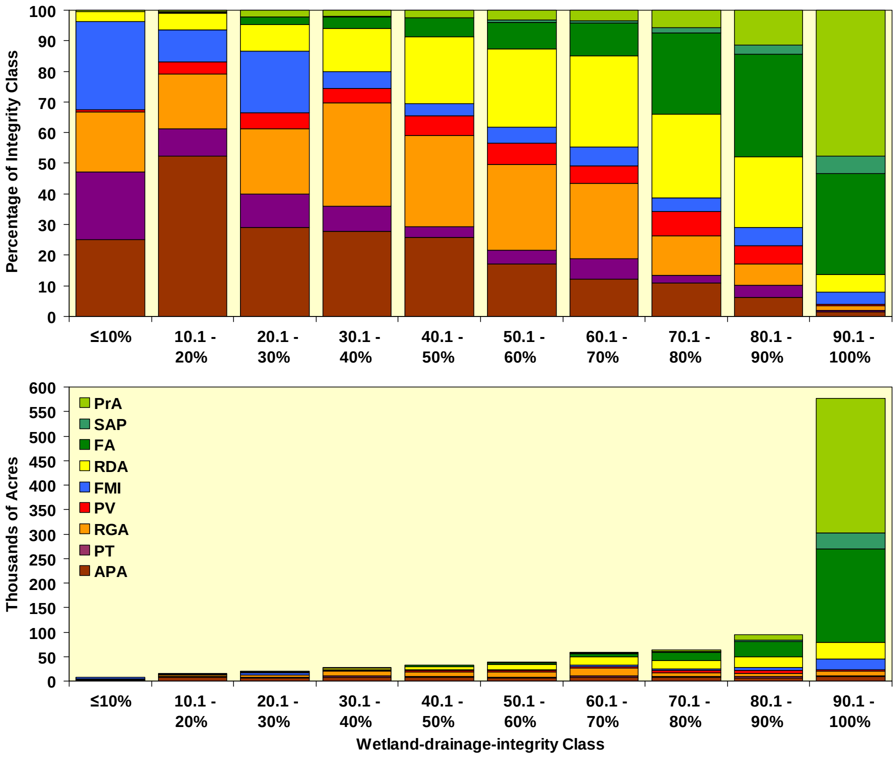

Figure 4.10. Pinelands management-area composition of wetland-drainage-integrity classes and the acres of drainage area in each class. The integrity values or scores were assigned to entire wetland-drainage units that include both habitat and non-habitat. Refer to Table 4.1 for Pinelands management-area codes.

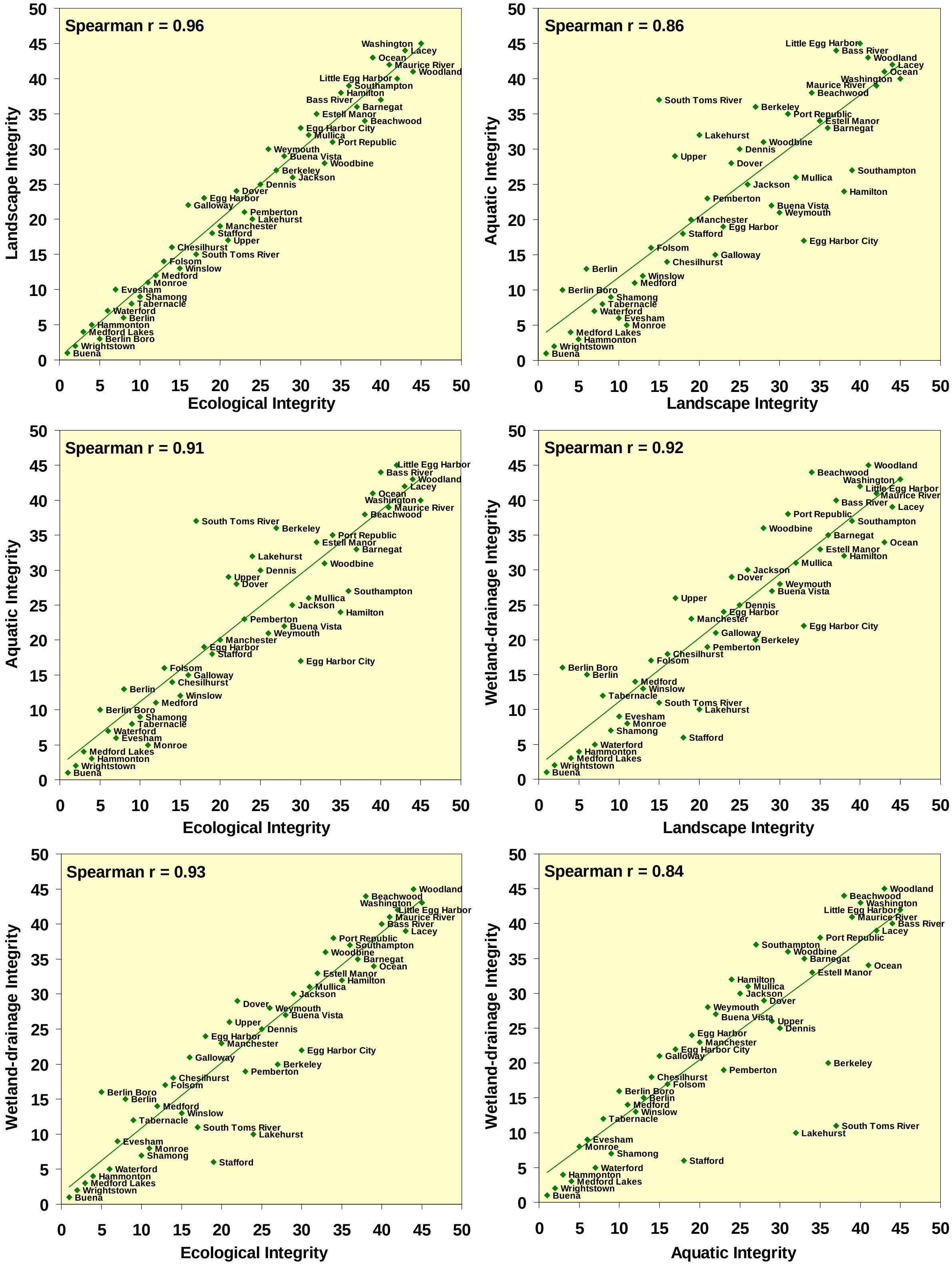

Figure 4.11. Relationship between the ecological-, landscape-, aquatic, and wetland-drainage integrity of Regional Growth Areas, Pinelands Towns, and Pinelands Villages (i.e., development areas) in Pinelands municipalities based on weighted-average ranks. Integrity increases along the x and y axes.

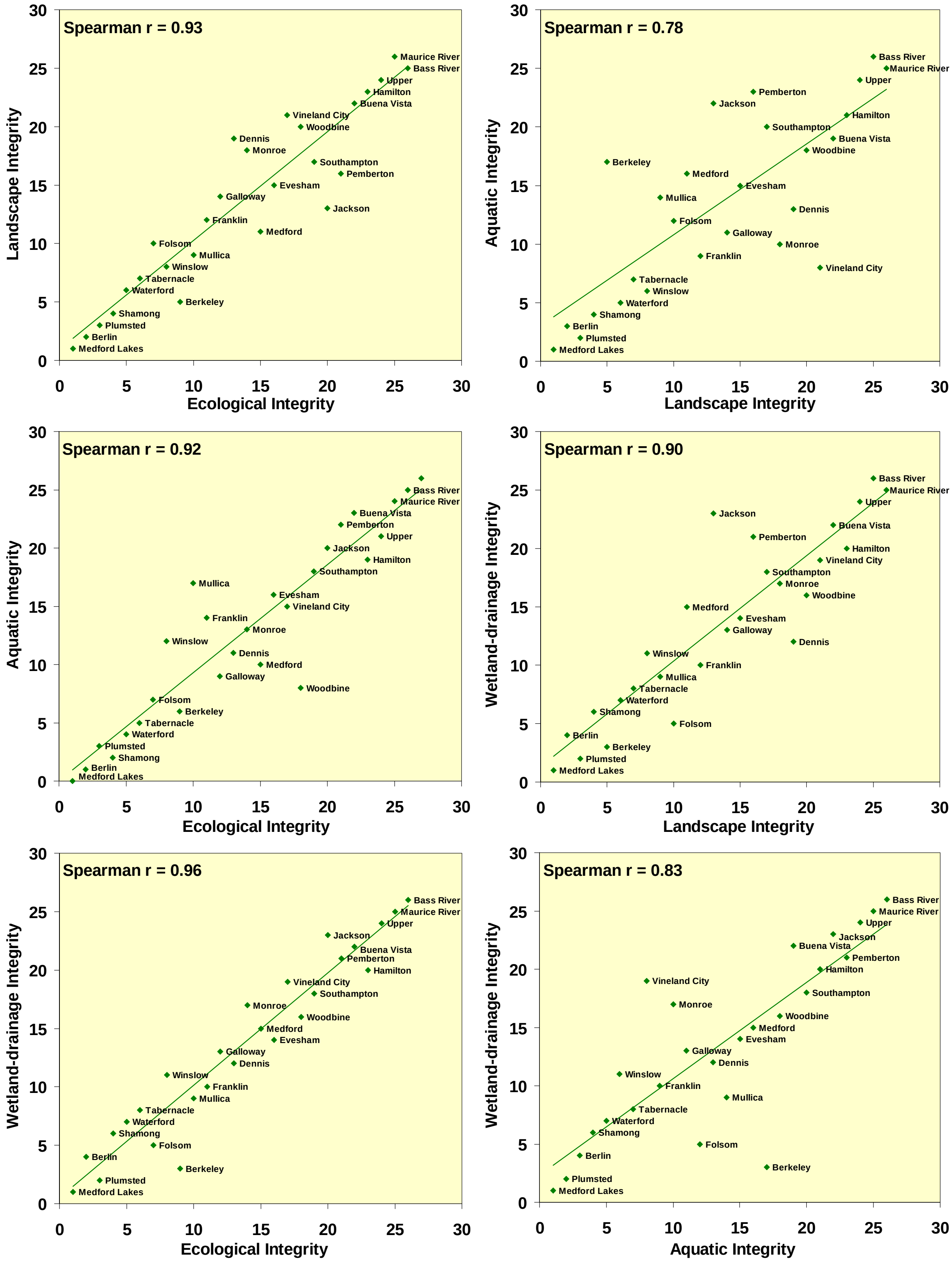

Figure 4.12. Relationship between the ecological-, landscape-, aquatic, and wetland-drainage integrity of Rural Development Areas in Pinelands municipalities based on weighted-average ranks. Integrity increases along the x and y axes.

Changes in Integrity from 1986-1995 and 1995-2002

Changes in landscape, aquatic, and wetland-drainage integrity that occurred from 1986-1995 and 1995-2002 included both increases and decreases in integrity (Tables 4.2-4.7). Changes in landscape integrity also included conversion to non-habitat. In each case, about 90% of the Pinelands Area remained unchanged. Areas of non-habitat were also reclassified as habitat of varying integrity. Changes in integrity were relatively small compared to the extent of the Pinelands Area that remained unchanged. Most losses of landscape, aquatic, and wetland-drainage integrity between periods were due to an area dropping to the next lowest integrity class (Tables 4.2-4.4).

The largest net increase in non-habitat during both periods occurred in Regional Growth Areas and Rural Development Areas (Figures 4.13). The area that displayed a decrease in landscape integrity was more than four times the area converted to non-habitat, suggesting that loss of habitat has a pronounced effect on the landscape integrity of remaining habitat. The greatest total net decrease in the integrity of Pinelands habitat in the top three landscape-integrity classes also occurred in these two management areas, with the greatest net decrease in the 90.1-100% class occurring in Forest Areas.

The greatest total net decrease in aquatic integrity in the top three classes occurred in Regional Growth Areas and Rural Development Areas, with the greatest net decrease in the 90.1-100% class occurring in Forest Areas (Figure 4.14). Although a net decrease in the 90.1-100% wetland-integrity-drainage class also occurred in Forest Areas and the Preservation Area, the greatest total-net decrease in wetland-drainage integrity in the 80.1-90% and 90.1-100% classes was associated primarily with Regional Growth Areas, Rural Development Areas, and Pinelands Towns (Figure 4.15).

Table 4.2. Landscape-integrity transitions for 1986-1995 and 1995-2005. Values are acres of Pinelands habitat and non-habitat. Values above the shaded-diagonal cells represent an increase in integrity between periods. Values below the shaded-diagonal cells represent a decrease in integrity between periods. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

		1995 Landscape-integrity class											
		Non-habitat	≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1986 Total
1986 Landscape-integrity class	Non-Habitat	156,482	-	87	237	292	649	671	932	1,264	1,914	1,513	164,041
	≤10	0	23	14	3	-	-	-	-	-	-	-	40
	10.1-20	30	5	698	165	-	-	-	-	-	-	-	898
	20.1-30	203	-	10	2,003	246	3	1	-	-	-	-	2,465
	30.1-40	307	-	-	241	4,706	464	<1	2	-	-	-	5,720
	40.1-50	509	-	-	<1	733	9,497	704	13	-	-	-	11,456
	50.1-60	1,079	-	-	<1	54	2,257	15,485	1,091	47	-	6	20,020
	60.1-70	2,093	-	-	-	130	394	4,577	28,449	1,734	106	43	37,526
	70.1-80	2,747	-	-	-	3	120	660	8,411	51,477	3,101	203	66,720
	80.1-90	3,697	-	-	<1	-	23	213	841	15,049	112,873	4,855	137,551
	90.1-100	2,413	-	-	-	-	<1	21	83	999	15,994	472,225	491,736
1995 Total		169,562	28	808	2,650	6,163	13,407	22,331	39,822	70,570	133,987	478,845	938,173

		2002 Landscape-integrity class											
		Non-habitat	≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1995 Total
1995 Landscape-integrity class	Non-Habitat	162,215	4	97	134	254	391	460	544	573	997	1,337	167,006
	≤10	9	20	6	-	-	-	-	-	-	-	-	34
	10.1-20	118	34	601	41	3	-	-	-	-	-	-	798
	20.1-30	222	-	138	2,074	167	5	-	-	-	-	-	2,607
	30.1-40	431	12	8	367	4,905	219	1	-	-	-	-	5,943
	40.1-50	611	-	1	2	1,117	10,458	437	-	-	-	-	12,627
	50.1-60	1,095	-	1	5	59	2,446	17,656	550	9	3	-	21,824
	60.1-70	1,984	-	-	5	124	516	5,492	31,273	676	5	6	40,079
	70.1-80	2,354	-	-	-	17	84	538	9,347	56,376	1,512	34	70,261
	80.1-90	2,563	-	-	-	-	1	136	574	10,460	113,780	3,056	130,570
	90.1-100	1,330	-	-	-	-	2	9	51	476	12,069	472,488	486,424
2002 Total		172,931	70	853	2,629	6,646	14,122	24,729	42,339	68,569	128,364	476,921	938,173

Table 4.3. Aquatic-integrity transitions for 1986-1995 and 1995-2005 given as acres of drainage units. Values above the shaded-diagonal cells represent an increase in integrity between periods. Values below the shaded-diagonal cells represent a decrease in integrity between periods. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

		1995 Aquatic-integrity class										
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1986 Total
1986 Aquatic-integrity class	≤10	2,939	128	0	0	0	0	0	0	0	0	3,067
	10.1-20	0	9,955	1,794	0	0	0	0	0	0	0	11,749
	20.1-30	0	4	14,440	1,053	0	0	0	0	0	0	15,498
	30.1-40	0	0	908	18,191	3,359	65	0	0	0	0	22,523
	40.1-50	24	0	0	2,868	26,090	667	0	0	0	0	29,650
	50.1-60	0	0	417	149	4,379	45,440	2,404	434	0	0	53,223
	60.1-70	0	0	0	440	1,070	14,255	54,165	1,567	85	246	71,827
	70.1-80	0	0	0	157	344	1,929	12,568	74,633	886	152	90,669
	80.1-90	0	0	0	0	9	175	793	21,201	129,843	3,107	155,129
	90.1-100	0	0	0	0	0	0	666	2,788	11,241	470,143	484,838
1995 Total		2,964	10,087	17,559	22,858	35,252	62,531	70,596	100,623	142,054	473,648	938,173

		2002 Aquatic-integrity class										
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1995 Total
1995 Aquatic-integrity class	≤10	2,886	49	0	0	0	0	0	0	0	0	2,935
	10.1-20	390	9,531	336	31	0	0	0	0	0	0	10,288
	20.1-30	417	853	14,804	1,239	0	0	0	0	0	0	17,314
	30.1-40	0	0	955	19,715	804	0	0	0	0	0	21,474
	40.1-50	0	0	198	4,472	28,480	41	0	18	0	0	33,208
	50.1-60	0	110	0	224	8,430	48,055	1,295	0	0	0	58,115
	60.1-70	0	0	149	40	361	14,761	56,123	875	1	57	72,366
	70.1-80	0	0	0	412	686	1,595	18,982	79,917	2,310	7	103,909
	80.1-90	0	0	0	0	114	321	664	19,933	116,788	1,881	139,700
	90.1-100	0	0	0	0	23	0	354	440	16,655	461,391	478,863
2002 Total		3,694	10,543	16,443	26,133	38,897	64,774	77,418	101,182	135,755	463,336	938,173

Table 4.4. Wetland-drainage-integrity transitions for 1986-1995 and 1995-2005 given as acres of wetland-drainage units. Values above the shaded-diagonal cells represent an increase in integrity between periods. Values below the shaded-diagonal cells represent a decrease in integrity between periods. Integrity classes are given as a range of percentages (e.g., 10.1-20%).

		1995 Wetland-drainage-integrity class										
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1986 Total
1986 Wetland-drainage-integrity	≤10	7,220	461	39	85	0	0	2	0	0	0	7,806
	10.1-20	367	11,228	2,017	16	0	39	0	0	0	0	13,667
	20.1-30	110	1,242	17,835	1,624	246	0	0	0	0	32	21,089
	30.1-40	2	272	1,339	19,022	2,143	203	115	0	0	17	23,112
	40.1-50	55	0	117	1,979	23,834	2,244	287	130	5	0	28,652
	50.1-60	0	63	873	582	3,413	26,965	2,448	403	50	5	34,802
	60.1-70	14	0	108	359	1,331	7,088	37,272	1,938	50	326	48,486
	70.1-80	0	0	0	551	803	703	12,957	47,259	3,341	424	66,039
	80.1-90	0	0	2	320	154	122	1,239	13,845	82,442	6,598	104,723
	90.1-100	73	56	56	37	293	322	678	2,031	15,374	570,876	589,797
1995 Total		7,841	13,321	22,386	24,577	32,217	37,688	54,999	65,606	101,261	578,277	938,173

		2002 Wetland-drainage-integrity class										
		≤10	10.1-20	20.1-30	30.1-40	40.1-50	50.1-60	60.1-70	70.1-80	80.1-90	90.1-100	1995 Total
1995 Wetland-drainage-integrity	≤10	6,626	1,031	3	9	0	0	71	0	0	0	7,740
	10.1-20	649	11,455	922	21	4	12	21	0	0	0	13,084
	20.1-30	424	2,953	16,938	1,227	0	12	0	0	49	19	21,623
	30.1-40	19	81	769	22,286	1,245	123	32	0	0	0	24,555
	40.1-50	63	92	463	4,157	23,966	1,524	349	0	0	7	30,622
	50.1-60	107	361	47	362	4,775	30,495	1,326	11	32	0	37,515
	60.1-70	0	0	258	280	1,930	4,746	45,235	1,441	102	109	54,101
	70.1-80	0	0	305	61	309	820	11,013	53,337	1,466	292	67,603
	80.1-90	0	0	0	10	386	763	540	8,366	81,907	4,535	96,507
	90.1-100	0	1	5	0	164	113	408	1,158	11,065	571,910	584,824
2002 Total		7,888	15,975	19,710	28,413	32,778	38,608	58,996	64,313	94,620	576,872	938,173

Table 4.5. Summary of landscape-integrity transitions for 1986-1995 and 1995-2002.

Landscape-integrity class	Status of 1986 habitat and non-habitat in 1995					Acres in each integrity class		Net change (1986-1995)	
	No change	Increased integrity	Converted to habitat	Decreased integrity	Converted to non-habitat	1986	1995	acres	%
Non-habitat	156,482	-	7,559	-	-	164,041	169,562	5,521	3.4
≤10%	153	115	-	-	-13	281	191	-90	-31.9
10.1-20%	983	139	-	-10	-94	1,225	1,371	145	11.9
20.1-30%	2,803	372	-	-108	-206	3,489	3,570	81	2.3
30.1-40%	5,198	914	-	-364	-364	6,839	7,256	416	6.1
40.1-50%	9,624	1,398	-	-1,106	-671	12,798	14,336	1,538	12.0
50.1-60%	17,115	2,149	-	-3,004	-1,418	23,686	26,212	2,526	10.7
60.1-70%	32,807	3,442	-	-6,304	-2,277	44,830	47,765	2,935	6.5
70.1-80%	51,085	4,493	-	-11,365	-3,123	70,066	71,019	953	1.4
80.1-90%	100,461	7,967	-	-15,456	-3,075	126,959	123,213	-3,746	-3.0
90.1-100%	464,579	-	-	-17,540	-1,839	483,959	473,677	-10,281	-2.1
1995 Total	841,290	20,987	7,559	-55,258	-13,080	938,173	938,173	0	0.0

Landscape-integrity class	Status of 1995 habitat and non-habitat in 2002					Acres in each integrity class		Net change (1995-2002)	
	No change	Increased integrity	Converted to habitat	Decreased integrity	Converted to non-habitat	1995	2002	acres	%
Non-habitat	162,215	-	4,792	-	-	167,006	172,931	5,925	3.5
≤10%	92	72	-	-	-36	200	138	-62	-30.9
10.1-20%	1,061	102	-	-28	-167	1,359	1,530	171	12.6
20.1-30%	2,879	163	-	-171	-249	3,462	3,732	270	7.8
30.1-40%	5,411	389	-	-554	-485	6,839	7,355	516	7.5
40.1-50%	11,224	605	-	-1,424	-768	14,021	15,835	1,814	12.9
50.1-60%	19,261	1,131	-	-3,489	-1,362	25,243	27,629	2,386	9.5
60.1-70%	36,770	1,869	-	-6,696	-1,973	47,309	49,098	1,790	3.8
70.1-80%	56,187	3,084	-	-10,783	-2,520	72,574	70,375	-2,199	-3.0
80.1-90%	104,538	5,291	-	-11,943	-2,208	123,980	119,355	-4,625	-3.7
90.1-100%	464,162	-	-	-11,069	-949	476,179	470,194	-5,986	-1.3
2002 Total	863,801	12,707	4,792	-46,157	-10,717	938,173	938,173	0	0.0

Table 4.6. Summary of aquatic-integrity transitions for 1986-1995 and 1995-2002.

Aquatic-integrity class	Status of 1986 drainage-unit area in 1995			Acres in each integrity class		Net change (1986-1995)	
	No change	Increased integrity	Decreased integrity	1986	1995	Acres	%
≤10%	2,939	128	-	3,067	2,964	-103	-3.4
10.1-20%	9,955	1,794	0	11,749	10,087	-1,662	-14.1
20.1-30%	14,440	1,053	-4	15,498	17,559	2,062	13.3
30.1-40%	18,191	3,423	-908	22,523	22,858	334	1.5
40.1-50%	26,090	667	-2,892	29,650	35,252	5,602	18.9
50.1-60%	45,440	2,838	-4,946	53,223	62,531	9,308	17.5
60.1-70%	54,165	1,898	-15,765	71,827	70,596	-1,231	-1.7
70.1-80%	74,633	1,038	-14,999	90,669	100,623	9,954	11.0
80.1-90%	129,843	3,107	-22,179	155,129	142,054	-13,075	-8.4
90.1-100%	470,143	-	-14,695	484,838	473,648	-11,190	-2.3
1995 Total	845,840	15,946	-76,387	938,173	938,173	0	0.0

Aquatic-integrity class	Status of 1995 drainage-unit area in 2002			Acres in each integrity class		Net change (1995-2002)	
	No change	Increased integrity	Decreased integrity	1995	2002	Acres	%
≤10%	2,886	49	-	2,935	3,694	758	25.8
10.1-20%	9,531	368	-390	10,288	10,543	254	2.5
20.1-30%	14,804	1,239	-1,271	17,314	16,443	-871	-5.0
30.1-40%	19,715	804	-955	21,474	26,133	4,659	21.7
40.1-50%	28,480	59	-4,670	33,208	38,897	5,689	17.1
50.1-60%	48,055	1,295	-8,765	58,115	64,774	6,659	11.5
60.1-70%	56,123	933	-15,310	72,366	77,418	5,052	7.0
70.1-80%	79,917	2,317	-21,675	103,909	101,182	-2,727	-2.6
80.1-90%	116,788	1,881	-21,030	139,700	135,755	-3,945	-2.8
90.1-100%	461,391	-	-17,472	478,863	463,336	-15,527	-3.2
2002 Total	837,691	8,945	-91,537	938,173	938,173	0	0.0

Table 4.7. Summary of wetland-drainage-integrity transitions for 1986-1995 and 1995-2002.

	Status of 1986 wetland-drainage-unit area in 1995			Acres in each integrity class		Net change (1986-1995)	
Wetland-drainage-integrity class	No change	Increased integrity	Decreased integrity	1986	1995	Acres	%
≤10%	7,220	586	-	7,806	7,841	35	0.4
10.1-20%	11,228	2,072	-367	13,667	13,321	-346	-2.5
20.1-30%	17,835	1,901	-1,352	21,089	22,386	1,298	6.2
30.1-40%	19,022	2,477	-1,613	23,112	24,577	1,465	6.3
40.1-50%	23,834	2,666	-2,152	28,652	32,217	3,566	12.4
50.1-60%	26,965	2,906	-4,932	34,802	37,688	2,885	8.3
60.1-70%	37,272	2,313	-8,901	48,486	54,999	6,512	13.4
70.1-80%	47,259	3,765	-15,015	66,039	65,606	-433	-0.7
80.1-90%	82,442	6,598	-15,683	104,723	101,261	-3,462	-3.3
90.1-100%	570,876	-	-18,921	589,797	578,277	-11,520	-2.0
1995 Total	843,953	25,284	-68,935	938,173	938,173	0	0.0

	Status of 1995 wetland-drainage-unit area in 2002			Acres in each integrity class		Net change (1995-2002)	
Wetland-drainage-integrity class	No change	Increased integrity	Decreased integrity	1995	2002	Acres	%
≤10%	6,626	1,114	-	7,740	7,888	149	1.9
10.1-20%	11,455	980	-649	13,084	15,975	2,891	22.1
20.1-30%	16,938	1,308	-3,378	21,623	19,710	-1,913	-8.8
30.1-40%	22,286	1,400	-869	24,555	28,413	3,858	15.7
40.1-50%	23,966	1,881	-4,776	30,622	32,778	2,156	7.0
50.1-60%	30,495	1,369	-5,651	37,515	38,608	1,093	2.9
60.1-70%	45,235	1,652	-7,215	54,101	58,996	4,895	9.0
70.1-80%	53,337	1,758	-12,507	67,603	64,313	-3,289	-4.9
80.1-90%	81,907	4,535	-10,066	96,507	94,620	-1,887	-2.0
90.1-100%	571,910	-	-12,914	584,824	576,872	-7,952	-1.4
2002 Total	864,153	15,995	-58,024	938,173	938,173	0	0.0

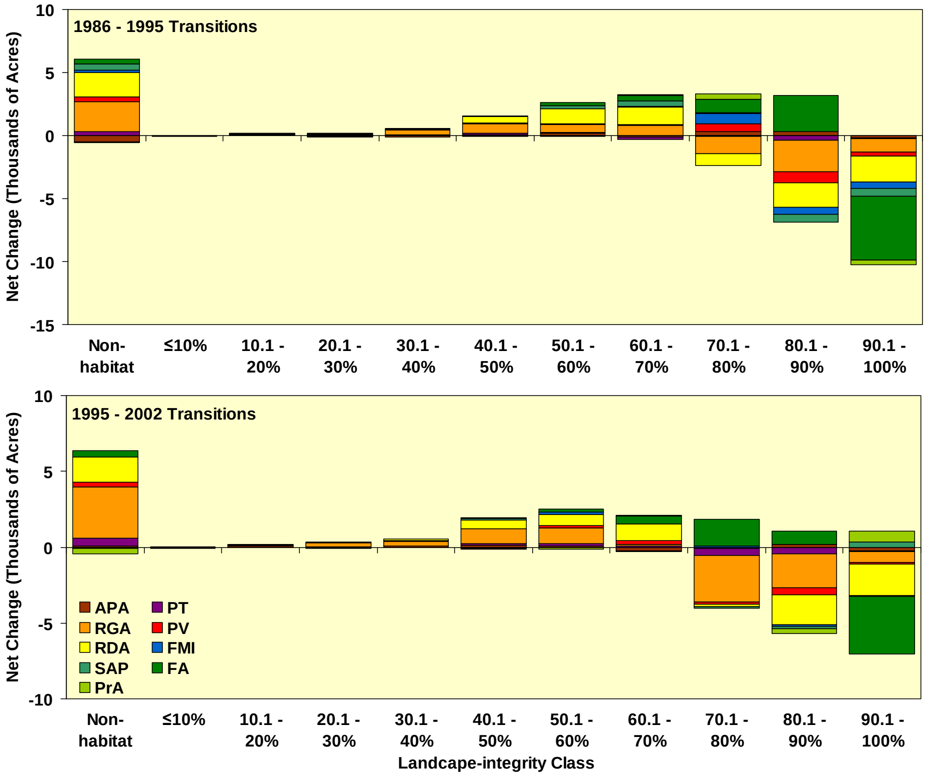

Figure 4.13. Net changes in the acres of habitat in each landscape-integrity class by Pinelands management area for the periods 1986-1995 and 1995-2002. Stacked bars show the nature of the change.

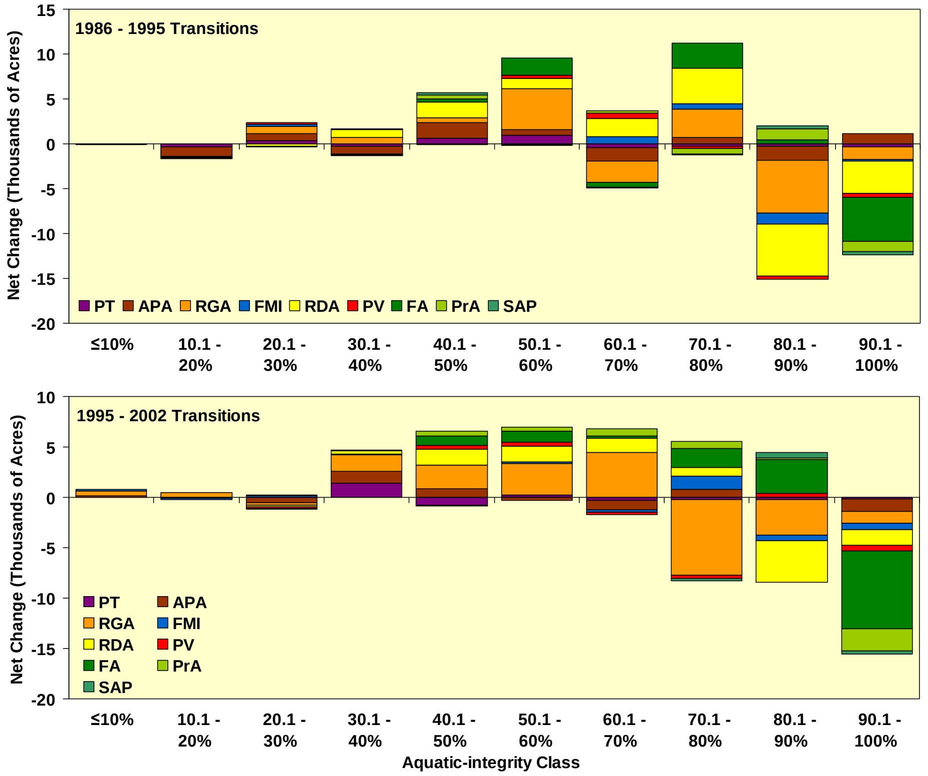

Figure 4.14. Net changes in the drainage-unit area in each aquatic-integrity class by Pinelands management area for the periods 1986-1995 and 1995-2002. Stacked bars show the nature of the change.

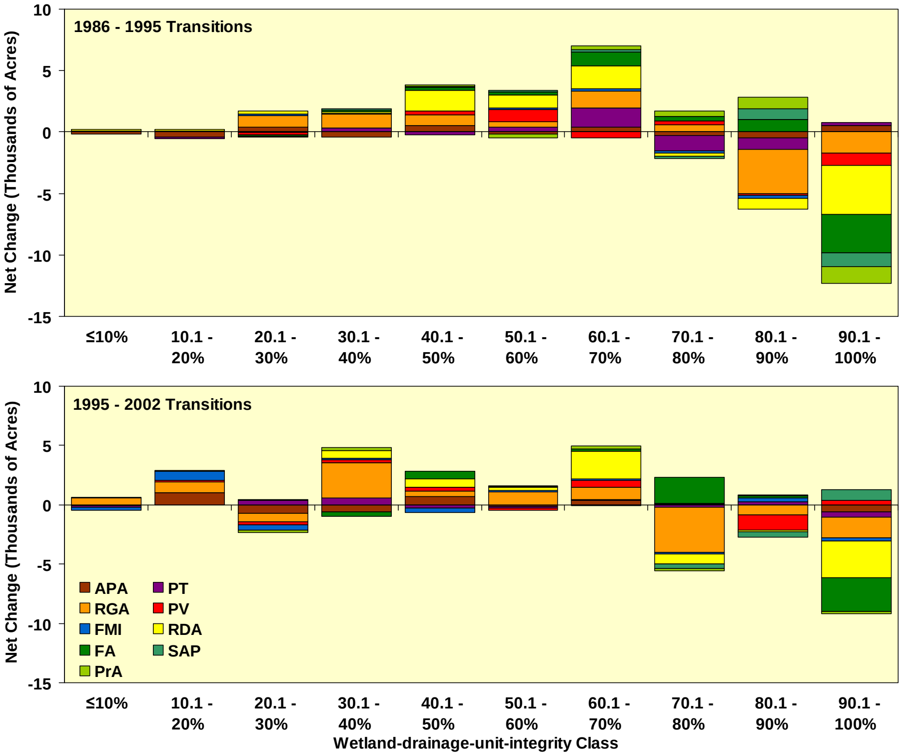

Figure 4.15. Net changes in the area of each wetlands-drainage-integrity class by Pinelands management area for the periods 1986-1995 and 1995-2002. Stacked bars show the nature of the change.

APPENDIX 5. EVALUATING WETLAND INTEGRITY

As indicated in the review of landscape, aquatic-community, and wetland studies (Appendix 1), habitat area, habitat loss, fragmentation, the intensity and proximity of adjacent land uses, and water-quality degradation influence the composition of both plant and animal communities associated with wetlands. Both the quantity and quality of groundwater flowing to most palustrine wetlands are influenced by land uses in adjacent uplands, whereas near-stream wetland communities can be affected by land use in the entire watershed. Many animal species range across upland and wetland landscapes, indicating that an assessment of Pinelands wetlands must also address the integrity of surrounding uplands. This is especially true for amphibians.

The ecological-integrity assessment, which integrates the results of the landscape-, aquatic-, and wetland-drainage-integrity assessments, provides a consistent, quantitative, and regional basis to evaluate the many ecological values associated with Pinelands wetlands. It also affords an opportunity to determine what percentage of wetland habitats falling within each integrity class might be affected by future land-use patterns.

The ecological-integrity assessment assigned integrity scores to individual 10×10-m Pinelands-habitat cells. A wetland complex within an individual wetland-drainage unit is composed of thousands of wetland-habitat cells that may display a range of ecological-integrity scores. A single score was assigned to discrete wetland units by averaging the ecological-integrity scores of all wetland cells within each wetland-drainage unit. Because the aquatic-integrity assessment provides a better characterization of conditions in streams, lakes, and other mapped water bodies, which cover 1.6% of the Pinelands Area, these habitats were not included in this analysis.

The results of the 2002 wetland-integrity assessment are presented spatially in Figure 5.1. Sixty-two percent of wetlands habitat in the Pinelands area fell within the highest wetland-integrity class (Figure 5.2). Employing the same weighted-average method used in the ecological-integrity assessment, each Pinelands management area (Table 5.1) was ranked based on its overall wetland integrity. For each of the nine management areas, the percentage of wetland habitat in each wetland-integrity class was multiplied by the upper range of the class (e.g., the weight for the 10.1-20% class equaled 20), the weighted percentages were summed and divided by 100, and the resulting management-area weighted-scores were ranked. The results indicated that wetland habitats in the Preservation Area District, Special Agricultural Production Areas, and Forest Areas displayed the highest overall wetland-integrity (Figures 5.3 and 5.4). These three management areas include 92% of the area in the 90.1-100% wetland-integrity class and 65% of the area in the 80.1-90% class. Wetland habitats in Pinelands Towns, Regional Growth Areas, and Agricultural Production Areas displayed the lowest overall wetland integrity, accounting for 70% of the wetland habitat in the five lowest integrity classes, with Agricultural Production alone accounting for 40% of the wetland habitat in these classes.

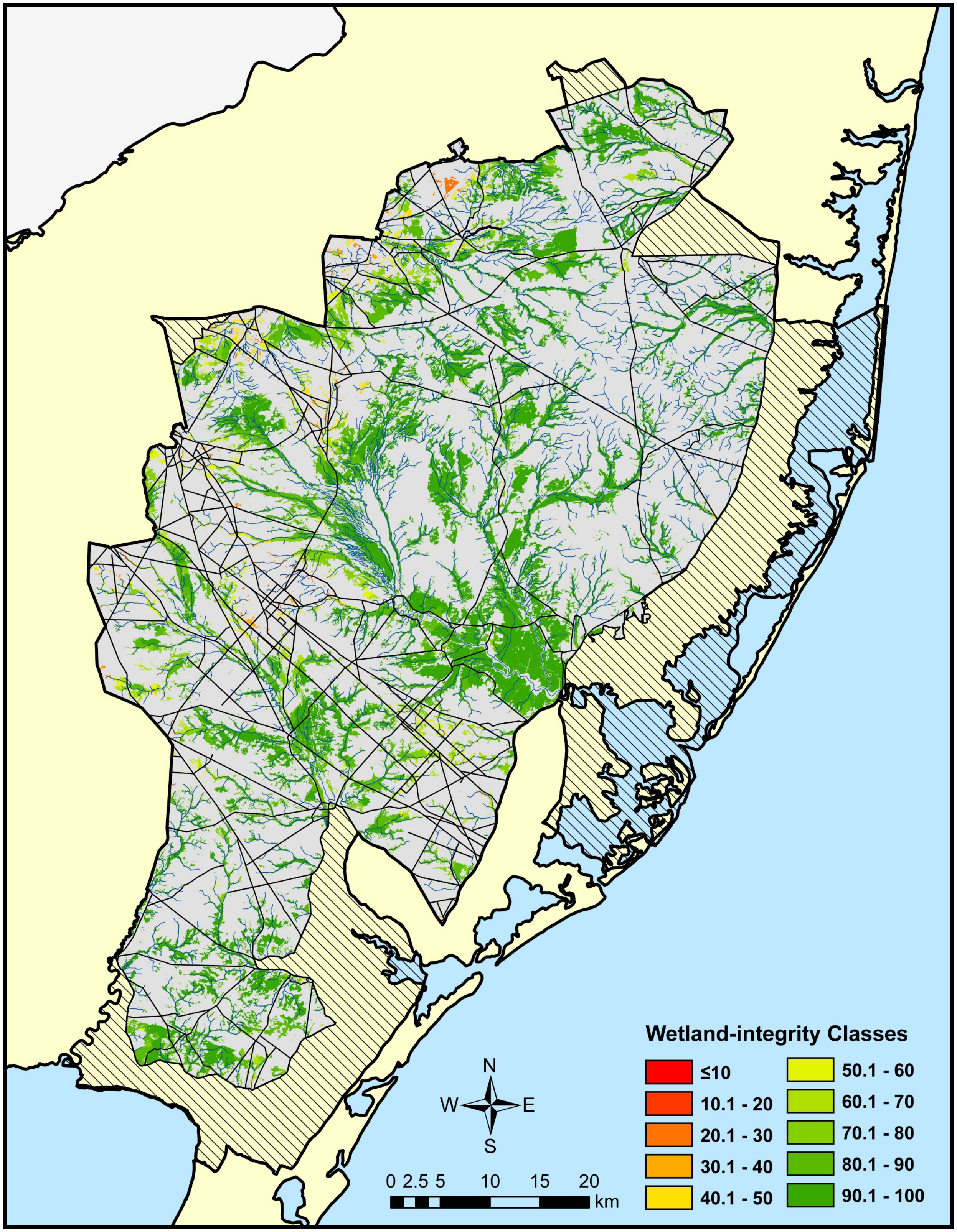

Figure 5.1. Wetland integrity. The wetland-integrity score assigned to discrete wetland units represents the average ecological-integrity score for all wetland-habitat cells within the associated wetland-drainage unit. The 90.1-100% class represents the highest level of ecological integrity. The hatched area shows the portion of the Pinelands National Reserve outside the Pinelands Area.

Table 5.1. Pinelands management areas. The area and the percentage of the Pinelands Area (%) that it represents are given for each management area.

Management area	Code	Acres	Hectares	%
Regional Growth Area	RGA	76,472	30,960	8.2
Pinelands Town	PT	21,758	8,809	2.3
Pinelands Village	PV	25,907	10,489	2.8
Rural Development Area	RDA	113,181	45,822	12.1
Federal and Military Installation Area	FMI	47,550	19,251	5.1
Agricultural Production Area	APA	68,160	27,595	7.3
Special Agricultural Production Area	SAP	37,582	15,215	4.0
Forest Area	FA	252,950	102,409	27.0
Preservation Area District	PrA	294,612	119,276	31.4
Pinelands Area	PA	938,173	379,827	100.0

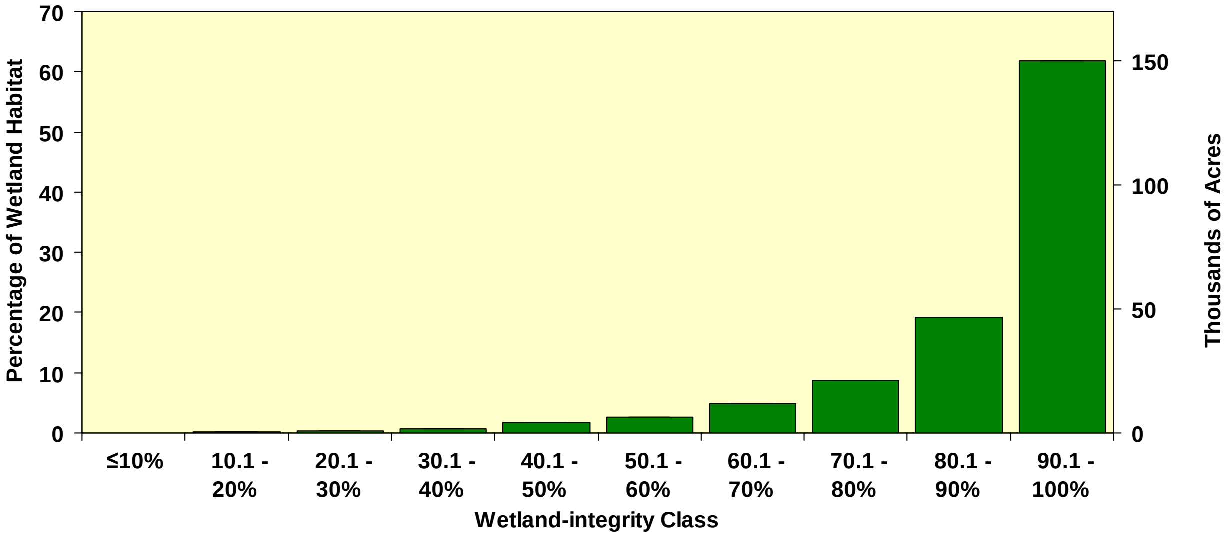

Figure 5.2. The percentage and acres of Pinelands wetland habitat within each of ten ecological-integrity classes ranging from ≤10% (lowest integrity) to 90.1-100% (highest integrity). Wetland integrity represents an average of the landscape-, aquatic-, and wetland-drainage-integrity scores assigned to discrete wetland units associated with wetland-drainage units.

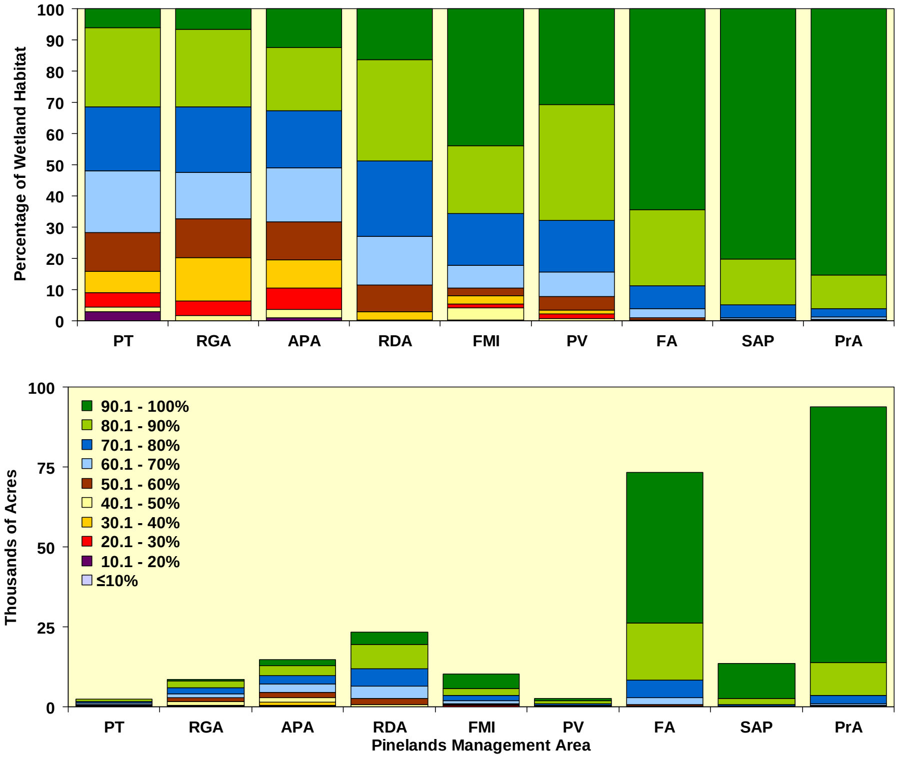

Figure 5.3. Wetland-integrity-class composition of wetland habitat by Pinelands management area. Refer to Table 1 for Pinelands management-area codes.

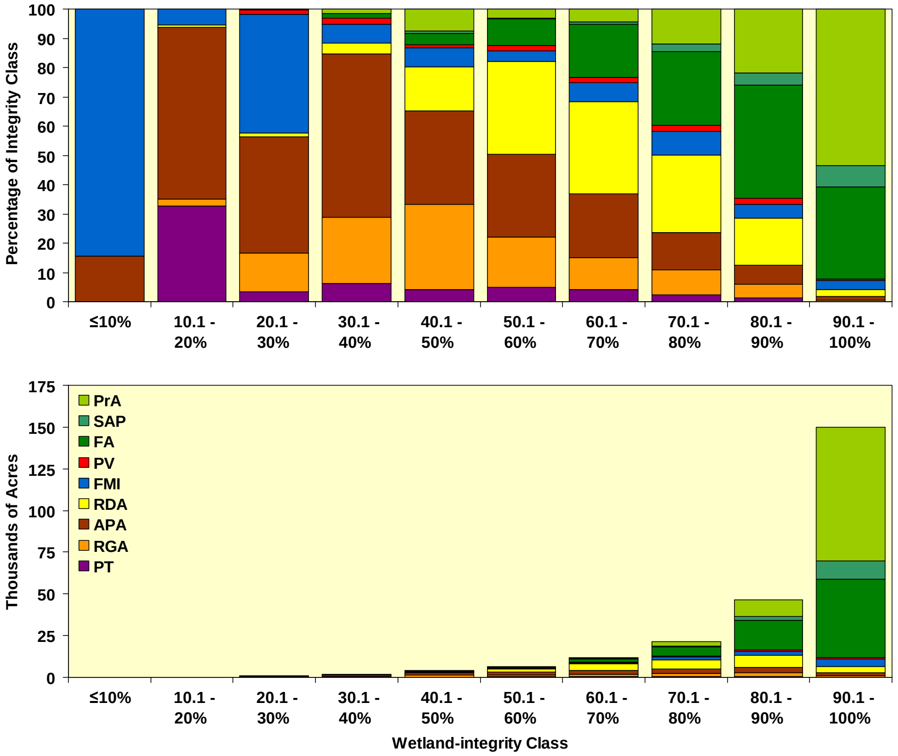

Figure 5.4. Pinelands management-area composition of wetland-integrity classes and the acres of wetland habitat in each class. Refer to Table 5.1 for Pinelands management-area codes.

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