Appendix: Conceptual Ecological Models

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Introduction

The development of conceptual ecological models which identify key system components, linkages and processes is a critical step in the design of a long-term monitoring program. The need for conceptual ecological models has been well established (Elzinga et al. 2001, Noon 2002), including by the NPS prototype park monitoring program. This experience demonstrates that conceptual models improve the planning process for monitoring by explicitly stating key elements of our understanding of system dynamics, which facilitates discussion, evaluation and refinement of the monitoring program (Maddox et al. 1999). Given the complexity of natural systems and the variety of factors that influence ecological processes, there is an obvious need for conceptual modeling as a tool to help organize information and synthesize understanding of system components and interactions. Failures in the development of major ecosystem monitoring programs have been attributed to the absence of sound conceptual models (NRC 1995).

The NPS Vital Signs Monitoring Program seeks to facilitate adaptive management by monitoring status and trends in 1) the ecological condition of park resources, 2) key anthropogenic stressors acting upon park systems, and 3) focal park resources. To accomplish this three-pronged objective, NETN has chosen to develop conceptual models which are both “effects-oriented” and a “predictive or stressor-oriented” (Trexler and Busch 2002). In other words, NETN conceptual models incorporate elements of ecological integrity, which integrate the effects of multiple drivers and stressors acting upon a system over time, as well as specific anthropogenic stressors and focal park resources.

A useful conceptual model or set of models for the NPS Vital Signs Monitoring Program should attempt to accomplish the following:

- Identify the bounds and scope of the system of interest
- Conceptualize and synthesize current understanding of system dynamics
- Identify major drivers and stressors acting upon the system and present our current understanding of system responses
- Integrate across disciplinary boundaries and spatial scales
- Describe and illustrate alternative hypotheses about key processes or system dynamics
- Aid in identifying appropriate indicators of ecological integrity and stressors
- Identify and illustrate key relationships among indicators and system dynamics
- Identify knowledge gaps which indicate the need for additional research
- Be updated as new information improves our understanding of the system
- Facilitate communication among scientists from different disciplines, managers, policy-makers, and the public

Given these lofty aspirations, it is important to remember that conceptual models are merely abstractions of our current understanding of the system. In reality, ecological systems are far too complex to be fully represented by our models. Moreover, these models must be flexible enough to allow change over time as our knowledge grows. Perhaps the most important characteristic of good conceptual modeling is the final point listed above: conceptual models foster communication and understanding among people with different backgrounds, goals, and points of view (Abel et al. 1998).

Conceptual Model Framework and Definitions of Components

Conceptual models may take the form of any combination of diagrams, narratives, tables, or matrices. In the development of conceptual models for the NETN Vital Signs Monitoring Program, we have chosen to employ both diagrammatic conceptual models, which help visualize system components and interactions, as well as narratives, which provide additional detail describing our current understanding of system components and interactions.

We have chosen a hierarchical approach to model development, beginning with a general model for the each of four key NETN ecological system groups (terrestrial, wetland, aquatic and intertidal). These general models identify key ecosystem drivers, stressors, ecological processes, elements of ecosystem condition, and focal park resources acting upon or present within each of these four major system groups. We present these general models, in the next four sections of this chapter, as diagrams accompanied by detailed narratives which lay out our current understanding of each of these components and their interactions.

A set of two diagrammatic models is then developed for each NETN park, which more specifically illustrate the specific stressors acting upon the ecological systems and aquatic resources, respectively, present within each park. These park models are included at the end of this Appendix. The aquatic park models include a hydrologic model of the freshwater inflows and outflows present in the park, as well as information describing freshwater resources. The aquatic models assume that ecosystem-wide processes such as precipitation and evaporation occur throughout the park, and that ground-water/surface-water interactions occur in both directions and also throughout the park.

This nested set of conceptual models incorporates multiple spatial scales that may be of interest to mangers. Landscape-, park-, stand- and species-level elements are all represented herein. Moreover, these models employ the tools and expertise of a range of scientific disciplines: landscape ecology, biogeochemistry, forest ecology, wetland ecology, aquatic ecology and conservation biology.

We have used the following terminology in developing NETN conceptual models:

Ecosystem Drivers

Ecosystem drivers are major external driving forces such as climate, hydrology, and natural disturbance regimes (e.g., hurricanes, droughts, fire) that have large-scale influences on natural systems.

Stressors

Stressors are physical, chemical, or biological perturbations to a system that are either foreign to that system, or natural to the system but applied at an excessive or deficient level (Barrett et al. 1976). Stressors cause significant changes in the ecological components, patterns and processes within natural systems. Examples include water withdrawal, pesticide use, timber harvesting, trampling, land-use change, and air pollution.

Focal park resources

Focal park resources are resources that, by virtue of their special protection, public appeal, or other management significance, have paramount importance for monitoring regardless of current threats or whether they are indicative of ecosystem integrity. Focal resources might include a species that is harvested, endemic, alien, or has protected status.

Indicators

Indicators are a subset of measurable ecosystem features or processes that are particularly information-rich in that their values are indicative of the quality, health, or integrity of the larger ecological system to which they belong (Noon 2002). Indicators are a select subset of the physical, chemical, and biological elements and processes of natural systems that represent the overall health or condition of the system.

Vital Signs

Vital Signs, as used by NPS, are indicators selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values. Vital signs may occur at any level of organization including landscape, community, population, or genetic level, and may be compositional (referring to the variety of elements in the system), structural (referring to the organization or pattern of the system), or functional (referring to ecological processes).

NETN Climate

Climate is a key ecosystem driver that affects the structure, composition and function of all ecological systems. The northeastern US has a temperate humid continental climate (Trewartha and Horn 1980); this climate displays large daily and seasonal temperature variation and abundant rainfall evenly distributed throughout the year (Bryson and Hare 1974). Temperature and rainfall vary latitudinally and altitudinally across the region. Mean annual temperatures range from about 11˚ C along the southern coast to 4˚ C in the northern highlands and annual precipitation ranges from 90-120 cm, of which from 10 to 30% falls as snow (Bryson and Hare 1974). The northern part of this region experiences cool summers, and long, cold winters which typically include a persistent snowpack from mid-December until April. In the southern part of this region, summers are warmer, winter temperatures are milder and snowpack development is more variable. The number of freeze-free days annually varies from only about 90 in the White Mountains to as many as 180 in a narrow strip along the southern coast (Bryson and Hare 1974). The climate of coastal regions is strongly influenced by the ocean; temperatures are more moderate and annual rainfall is slightly higher along the coast, and summer fog is common along the Maine coast (Bryson and Hare 1974). The Northeastern U.S. is in the path of many frontal systems; these typically move eastward across the continent until reaching the Atlantic Ocean then travel northeastward along the coast. Low pressure cells in the frontal systems generate counterclockwise winds that being warm, moist air from the Atlantic Ocean onto the mainland. Rain or snow is released when a warm airmass meets a cold front (Maloney and Bartlett 1991).

Terrestrial Resource Conceptual Model

We have developed a conceptual ecological model to identify key ecosystem drivers, stressors, ecological processes, elements of ecosystem condition, and focal park resources acting upon or present within each of four general ecological system groups present within NETN. Herein, we present the terrestrial resource conceptual ecological model as a diagram (Figure 1) accompanied by the following narrative describing our current understanding of these components and their interactions.

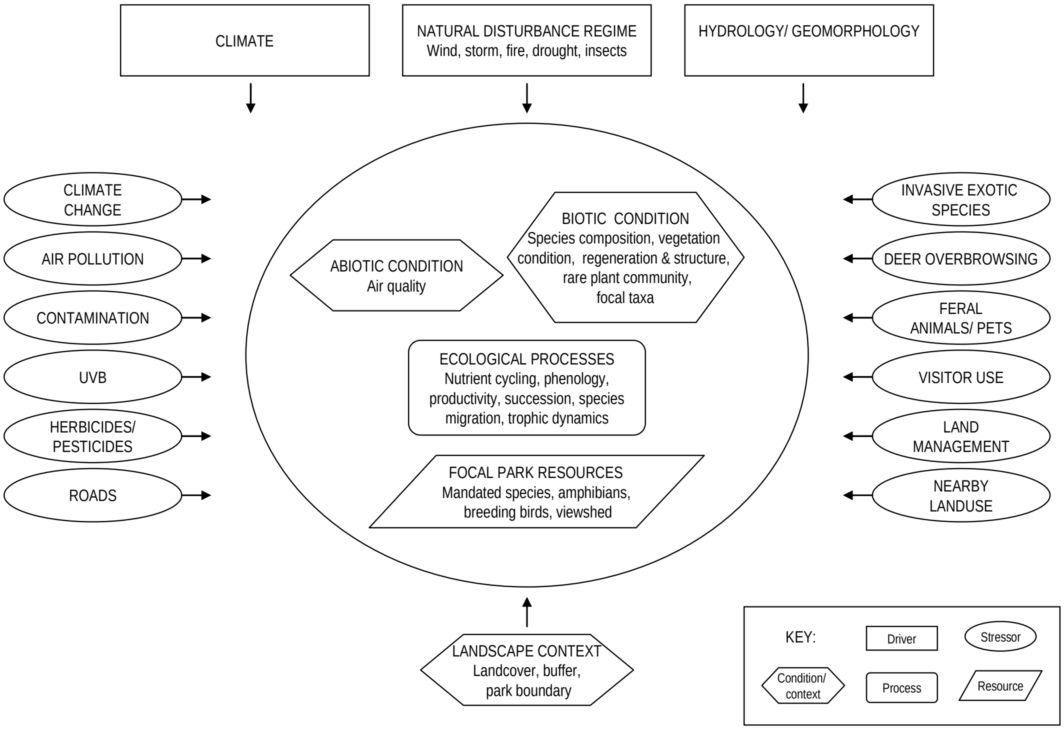

Figure 1. Terrestrial conceptual model diagram.

<u>Ecological systems</u>

Terrestrial ecological systems present within NETN parks encompass a variety of forested systems and several types of open uplands and human-modified systems (Table 1, Figures 5, 7, 9, 11, 13, 15, 17, 19, 21, 23). Within the northeastern United States, a temperate climate with abundant rainfall acting on gently rolling and occasionally mountainous topography upon mostly acidic bedrock and glacial till creates suitable conditions for a variety of terrestrial vegetation. The topography and ecology of this region reflects its glacial history, which left a varied landscape of lakes, depressions, morainic hills, drumlins and other glacial features. While fine scale variation in site conditions and natural disturbance create a diverse patchwork of varied forest associations, broad scale patterns are evident. Latitudinal and altitudinal variation in temperature, soil quality and disturbance regimes from the coast up into the mountainous regions of New Hampshire, Vermont, and western Massachusetts create the broad ecological system groups described below.

Table 1 Natureserve ecological systems present within NETN parks, and approximate hectares of each system within each park. This information will be updated as better information becomes available after completion from the mapping inventory phase of the I&M program within these parks.

	Ecosystem Category	Ecological System Type	NatureServe Code	Hectares of system within park									
				ACAD	BOHA	MABI	MIMA	MORR	ROVA	SAGA	SAIR	SARA	WEFA
Terrestrial uplands	Spruce-fir forest	Acadian Lowland Spruce-Fir-Hardwood Forest	CES201.565	6588									
	Northern hardwoods/mixed forest	Northern Aspen-Birch Forest	CES103.020	1160									
		Laurentian-Acadian Northern Hardwoods Forest	CES201.564	314		33	20		83	13		273	
		Laurentian-Acadian White Pine-Red Pine Forest	CES201.719	737									
		Laurentian-Acadian Pine-Hemlock-Hardwood Forest	CES201.563	881		97			77	19		432	
		Appalachian Hemlock-Hardwood Forest	CES202.593										
	Central hardwoods forest	Central Appalachian Oak and Pine Forest	CES202.591			1		229					12
		Northeastern Interior Dry Oak Forest	CES202.592		+		112		3				
		Central and Southern Appalachian Northern Hardwood Forest	CES202.029					44					
	Open uplands	Laurentian-Acadian Acidic Rocky Outcrop	CES201.571	3295									
		Laurentian-Acadian Alkaline Rocky Summit	**CES201.XXX**						0.04				
	Cliff and talus	Laurentian-Acadian Calcareous Cliff and Talus	CES201.570			0.2							
		Laurentian-Acadian Acidic Cliff and Talus	CES201.569	11									
	Rocky shore	Acadian-North Atlantic Rocky Coast	CES201.573	116	+								
	Modified	Native Plantation				45			2	11		4	
		Exotic Plantation				18			12				
		Old-field successional				3	62	193	15			162	
		Open fields				17		24		6			4
		Agricultural fields					42		8			206	
		Landscaped grounds		112		4	14		50	2	1	5	
Freshwater wetlands	Floodplain forest	Laurentian-Acadian Floodplain Forest	CES201.587				4					30	
		Central Appalachian Floodplain	CES202.608										
	Softwood/hardwood swamp	Laurentian-Acadian Alkaline Swamp	CES201.575										
		Laurentian-Acadian Acidic Swamp and/or Eastern Boreal Semi-Treed Bog	CES201.574 and/or CES201.581	292									
		Laurentian-Acadian Acidic Swamp	CES201.574	68		4	58					3	
		North-Central Appalachian Acidic Swamp	CES202.604					6	9				1
	Peatlands	Acadian Maritime Bog	CES201.580	37									
		Laurentian-Acadian Acidic Basin Fen	CES201.583	174			2						
		Laurentian-Acadian Alkaline Fen	CES201.585	68									
	Wet meadows/shrub swamps	Laurentian-Acadian Wet Meadow-Shrub Swamp and Marsh	CES201.577	223	+	0.2	42	9	3	5		4	4
		Laurentian-Acadian Wet Meadow-Shrub Swamp and Marsh and/or Laurentian-Acadian Acidic Basin Fen	CES201.577 and/or CES201.583	43									
Tidal & intertidal wetlands	Coastal marsh	Acadian Coastal Salt Marsh	CES201.578	38									
		Acadian Estuary Marsh	CES201.579		+								
	Rocky shore	North Atlantic Cobble Shore	CES201.051		+								
		North Atlantic Rocky Intertidal	CES201.048		+								
	Mud and sand flats	North Atlantic Intertidal Mudflat	CES201.050		+								
		North Atlantic Tidal Sand Flat	CES201.049		+								
	Unspecified tidal zone			81							2		

Forested ecological systems within NETN parks can be divided into four general groups (Westveld 1956, Foster 2004): 1) the Central hardwood forest of southern New England and parts of New Jersey and New York, dominated primarily by oaks with other hardwood species; 2) the Transition hardwood forest of central New England, in which oaks and hickories of the central hardwood forest intermingle with northern hardwood species as well as hemlock and white pine; 3) the Northern hardwood forest of northern New England, dominated by American beech, yellow birch and sugar maple, with a variety of other hardwood species and hemlock and white pine; and 4) the Spruce-fir forest found at higher elevations in northern New England and along the Maine coast, dominated by red spruce and balsam fir, with white and black spruce. The Spruce-fir forest is often replaced after major disturbance by a Boreal aspen-birch forest.

Relatively open uplands occur within NETN at high elevations, upon steep slopes and along the coast. These systems are primarily present along the APPA and at ACAD. These open uplands can be classified generally into four categories: 1) oak and pine woodlands and shrublands, often accompanied by low heath shrubs, which occur on and around rocky summits; 2) alpine barrens which occur above treeline along the APPA and are characterized by dwarf shrubs, lichens, and areas of sparse vegetation; 3) sparsely vegetated cliff and talus systems which develop near steep cliffs and include the dry, exposed talus slopes that develop beneath the cliffs; and 4) patchy shrub vegetation which develops within the narrow strip just above the high tide line along the New England rocky coast, where tree growth is inhibited by wind, salt spray and fog.

Several human-modified terrestrial habitat types occur within NETN parks. Native and/or exotic tree plantations have been established at MABI, ROVA, and to a lesser extent, at other parks. Successional old-field, maintained open fields and active agricultural fields are all present within the historic parks, which are managed to perpetuate historical landscapes. Finally, some areas of landscaped grounds are present within most NETN parks.

The presence of one rare terrestrial community type has been confirmed at ACAD; the Pitch pine/Broom crowberry woodland is ranked G1/G2 by NatureServe, indicating this community type is globally imperiled or critically imperiled.

Ecosystem Drivers

Climate

Climate is a key ecosystem driver that affects ecosystem structure, composition and function. The climate of the northeastern US is described above. Temperature and rainfall vary latitudinally and altitudinally across the region creating gradations between the Central and Northern hardwood and Spruce-fir forests, and the open and alpine uplands. The potential and realized impacts of changing climate upon terrestrial systems within NETN parks are described below within the section describing Stressors.

Disturbance Regimes

Disturbance regimes are a second key driver affecting NETN terrestrial systems. The forces of disturbance work upon this forested landscape to create a shifting mosaic of forest regeneration and succession. Throughout the entire region, frequent windstorms create small to medium size gaps that rapidly regenerate; these windstorms are more frequent along the coast and on windward slopes (Lorimer and White 2003). Less frequent hurricanes create much larger openings and temporarily create habitat for earlier successional species within the forest mosaic. Periodic ice storms can cause substantial damage over large regions, but tend to result in regeneration rather than stand replacement (Lorimer and White 2003). The natural role of fire within this region is complex and less well understood. Historically, fire has been infrequent within the northern hardwoods forest, but was more common within the central hardwoods forest and probably also within the transitional mixed forest between northern hardwoods and spruce-fir (Cogbill et al. 2002). At the time of European settlement, the central hardwood oak forests of the southern New England coast were more open forests with sparse understory, perhaps due to propagation of fire by Native Americans (Cogbill et al. 2002). European settlement drastically increased the frequency and intensity of fires throughout the northeastern US, particularly due to logging and railroad traffic during the 19th century. Much of Acadia National Park burned during a historic fire in 1947. Modern fire suppression regimes have now essentially removed fire as an agent of disturbance in this region. Insect pests and disease are also important agents of natural disturbance, particularly in the low diversity coniferous forest. The native spruce budworm has periodically killed large numbers of mature balsam fir which then blowdown or burn creating large-scale disturbance (Lorimer and White 2003).

Disturbance regimes within terrestrial systems are best considered at multiple scales. Infrequent, larger-scale disturbance is evident at the scale of the landscape and can be monitored using remote sensing. More frequent, smaller-scale disturbance is evident at the scale of the stand and can be monitored on-site. The potential and realized impacts of altered disturbance regimes due to climate change and invasive exotic species are discussed below within the section describing Stressors.

Ecosystem condition

Within terrestrial ecosystems, the structure, composition and condition of dominant vegetation play a key role in determining ecosystem function and quality of habitat for other species. For this reason, monitoring key elements of vegetation structure, composition and condition are particularly important elements for long-term monitoring. In addition, NETN has attempted to identify taxa that may be useful indicators of the condition of particular functional or taxonomic groups or of response to specific stressors. While the use of specific taxa as indicators of ecological condition is controversial (Prendergast et al. 1993), this approach can be useful if a range of species representing various taxa and life histories can be included (Terborgh 1974, Griffith 1997, Carignan and Villard 2002). By monitoring diverse taxa, we reduce the chance of failing to detect significant change in the ecological integrity of these systems. In theory, it is attractive to select a parsimonious set of “best” indicators of ecological condition for these systems; in reality, we must realize that our current understanding of ecological systems is still quite limited, and that a narrowly focused monitoring program will fail to detect significant change within many important but less-charismatic taxa.

Vegetation community structure and demography

Vegetation community structure and demography are fundamental properties of terrestrial ecosystems. Monitoring the composition and structure of forest communities provides basic information on changes in forest cover type, species composition, and the type and quality of available wildlife habitat; moreover this basic information will allow proper interpretation of many other Vital Signs. Monitoring vegetation demography in the form of tree seedling and sapling regeneration provides an anticipatory indicator of future forest cover type as well as an integrative measure of the impacts of multiple stressors acting upon vegetation. Monitoring canopy and understory tree mortality provides another key integrative measure of multiple stressor impacts. Stand structural or age class is indicative of both successional stage and habitat quality, and is a particularly useful measure in forest systems subject to silviculture. Legacy features, such as large trees, snags and coarse woody debris provide important habitat for birds, mammals, and herptiles, as well as decomposers, bryophytes and tree seedlings. These legacy features can be useful indicators of wildlife habitat within early- and mid-successional forests and those subject to silviculture.

Canopy vegetation condition is an integrative, anticipatory indicator of stress and change within canopy vegetation, which can in turn lead to changes in ecosystem function, habitat quality and stand composition. Canopy condition can be affected by a multitude of drivers and stressors, including climate, outbreaks of insect pests or disease, atmospheric deposition, tropospheric ozone pollution, and nutrient availability (Bonneau 1999, Shugart et al. 2000). Canopy vegetation condition can be measured across the landscape using vegetation stress indices from hyperspectral remote sensing (Sampson et al. 2000, Miles et al. 2003); while hyperspectral imagery is currently expensive to obtain, this technology is advancing rapidly and should be considered for inclusion in the NETN monitoring program as affordable imagery becomes available. At the stand scale, canopy condition can be assessed visually onsite as the crown condition of each canopy tree in a plot.

Selection of taxonomic groups as indicators for long-term monitoring should, to the extent possible, detect response to a wide range of stressors at several spatial scales (Noss 1990, O'Connell 1998), and include the range of functional and taxonomic groups important in a particular ecosystem (Keddy and Drummond 1996). Within temperate forested ecosystems, mycorrhizal fungi, and arthropod pollinators and decomposers, are key taxa performing essential ecosystem functions, while forest herbs and native avian communities are highly visible "flagship" taxa which are also sensitive to a variety anthropogenic stressors. Ectomycorrhizal fungal communities have been shown to be sensitive to nitrogen deposition, a key component of acidic deposition (Arnolds 1991, Lilleskov and Bruns 2001). While short-term trends in ectomycorrhizal sporocarp emergence exhibit substantial temporal variation due to climatic fluctuations and other factors, long-term data collection may allow the elucidation of trends in the relative change in functional groups of ectomycorrhizae from sporocarps (Lilleskov, personal communication). Avian communities may be particularly well-suited to long-term monitoring due to their sensitivity to habitat fragmentation, and the ease of avian identification using vocalization (Carignan and Millard 2002).

Selected arthropod taxa could provide useful indicators of environmental condition at the scale of the park. In general, arthropod populations inhabit smaller home-ranges than many larger and more charismatic fauna, and thus may be more useful as indicators of environmental condition within these relatively small parks. Moreover arthropods typically have shorter lifespans than larger faunal taxa; this may increase their utility as "early-warning" indicators. By monitoring taxa with specific functional relevance, such as pollinators and decomposers, NETN could incorporate these important ecological processes within NETN ecological condition ratings. Studies have shown that carrion beetles are sensitive to forest fragmentation (Gibbs & Stanton

2001); these beetles are important decomposers, and are straightforward to monitor. Honeybees are another taxa with important functional relevance that may be feasible for inclusion in a long-term monitoring program (Sam Droege, personal communication).

One additional species for inclusion in a long-term monitoring program is the red-backed salamander, a species which comprises a significant component of faunal biomass within temperate forested systems, in which it is widely distributed. This species has been monitored as an indicator of acid stress and climate change (Welsh and Droege 2001). Long-term monitoring of this suite of taxa would provide data indicative of the condition of a broad range of taxa with functional significance and high-value to humans.

In addition to forested ecosystems, NETN parks contain substantial areas of open field and successional old-field habitat, which is maintained within many NETN historic parks to satisfy cultural mandates. While highly modified, these systems provide important habitat for many grassland and shrubland species, such as the upland sandpiper, Henslow's sparrow, grasshopper sparrow, savannah sparrow, bobolink and Eastern meadowlark (Bernardos et al. 2004). These birds proliferated within the agrarian landscape of nineteenth and early 20th century New England, and populations of these birds have declined significantly as the landscape has reforested over the last century; thus the national historic parks provide essential habitat for these vulnerable species.

Ecological process

Nutrient Cycling

Nutrient cycling is a fundamental ecological process that is intrinsically linked to the composition, productivity and function of ecosystems. The utility of using some measures of nutrient cycling as indicators of ecosystem status, function or integrity has been widely recognized (Harwell et al. 1999, The Montreal Process 1999). Plant growth in northeastern forested ecosystems has historically been limited by nitrogen, an essential macronutrient in limited supply. Anthropogenic atmospheric deposition of nitrogen compounds has altered patterns of nitrogen cycling in northeastern forests, increasing both the supply of available nitrogen and nitrate loss from the system, a trend which may lead to nitrogen saturation and increased acidification (Aber et al. 1998). Anthropogenic atmospheric deposition has also caused the loss of essential base cations from terrestrial systems via leaching, and has increased availability within the soil of aluminum – a phytotoxin (Driscoll et al. 2001). Monitoring a few simple measures of terrestrial N cycling (nitrification, soil C:N ratio) and base cation and aluminum availability (soil base saturation, soil Ca:Al ratio) will provide useful information indicating the level of stress from atmospheric deposition experienced by forested systems (Driscoll et al. 2001). Additionally, monitoring of nitrate in streamwater across the landscape will allow some assessment of patterns and degree of nitrogen saturation.

Productivity

The productivity of a natural ecological system provides a measure of energy flow through the system; productivity is the amount of energy stored as organic matter. Patterns of productivity are strongly dependent upon temperature, rainfall, and solar radiation, thus productivity varies with vegetation physiognomy. Within an ecological system, annual productivity varies with climate and patterns of disturbance as well as with stressors such as insect or herbivore browsing and atmospheric deposition and ozone (Laurence and Anderson 2003, Ollinger et al. 2002). Thus productivity provides an integrated measure of the status of an ecological system or of a specified taxa. Forest productivity can be measured via remote sensing using indices of chlorophyll concentration (Sampson et al. 2000, Smith et al. 2002), or from stand measurements of forest growth.

Phenology

Northeastern temperate systems are characterized by distinct seasonality that drives patterns of plant and animal phenology. Recent research indicates that anthropogenic climate change may already be driving phenological change in a variety of species (Root et al. 2003, Parmesan and Yohe 2003). Monitoring key phenological occurrences such as bud break and flowering in key species will help determine the magnitude and patterns of such change within NETN systems.

Landscape context

The landscape of New England has been profoundly altered by human activities over the last four hundred years (Foster et al. 2004). Widespread forest-clearing for agriculture and logging for timber have left very few forested stands in the northeastern US untouched. In particular, the southern New England coast and adjacent areas of New York and New Jersey are among the most densely settled areas within the US, resulting in the elimination or drastic alteration of all of the central hardwoods forest within this region. Remaining areas are small, fragmented and heavily impacted by human activities. Larger areas of northern hardwood forest, and spruce-fir forest remain within northern New England. Most of these areas were logged, or cleared and plowed during the 19th century, and some have now returned to “mature” forest that in some ways resemble pre-settlement forest, while others are currently managed for timber production. Remaining forested areas exist in a matrix of managed, rural, and suburban habitat that limit the ability of species to interbreed and disperse, and introduce “edge effects.” A network of roads cuts through northeastern forested regions, reinforcing edges and introducing disturbance, pollutants, de-icing chemicals, and facilitating invasion by exotic species (Brothers and Spingam 1992, Spellerberg 1998). Compared to forest interiors, forest edges are windier, subject to greater extremes in temperature, are more accessible to specific predators, and receive higher loads of some atmospheric pollutants (Harrison and Bruna 1999, Weathers et al. 2001). A large and growing body of scientific literature documents the negative impacts of habitat fragmentation on biodiversity in a wide variety of ecological systems (Fahrig 2003). The impacts of fragmentation have been especially well documented upon avian communities and population declines of a variety of forest interior avian species is linked to habitat fragmentation (Rich et al. 1994, Austen et al. 2001, Boulinier et al. 2001). Habitat fragmentation is increasingly being monitored using remote sensing, and a wide variety of indices are available to describe patterns of fragmentation.

Stressors

Today, the forests of New England are subjected to a suite of anthropogenic stressors unlike anything encountered during their long history prior to European settlement. These stressors act as agents of change in a myriad of related and often interacting ways. While the effects of some stressors, like acidic deposition, have been extensively studied and are well understood (Driscoll et al. 2001), the effects of other important stressors, like climate change, are complex and unpredictable enough to elude our understanding despite concerted and ongoing study (McNulty and Aber 2001). The impacts of still other stressors, such as many newly invading species, are yet to be studied. The impacts of many stressors will vary depending upon landuse history (Foster et al. 2003), and the combined impact of this suite of interacting stressors is certain to yield unexpected results (Aber et al. 2001).

Invasive Species

The effects of invasive exotic species on the structure, composition and function of natural systems have become a chief concern of ecologists and land managers over the last 20 years (Drake et al. 1989). Spread of many invasive species is aided by disturbance and may increase as anthropogenic disturbance of native ecosystems continues to increase.

Currently, northeastern forests are being seriously impacted by several species of invasive exotic insect pests and pathogens. The hemlock wooly adelgid has caused widespread mortality of hemlock across the eastern US since introduction here in the 1950s, and threatens to rapidly and substantially reduce or eliminate eastern hemlock throughout much of its range (Orwig et al. 2002). Infestation currently extends into southern New England, but isolated occurrences further north indicate this pest will continue to spread. Only slightly less serious is the threat to American beech. An exotic scale insect has caused the widespread occurrence of beech bark disease throughout the northeastern forests. Caused by the interaction of this insect and a native fungus, beech bark disease has caused substantial beech mortality throughout the region, though most immature and some mature trees have some resistance to the disease. In areas of high beech mortality, increased sprouting of beech suckers can dramatically alter forest structure (LeGuerrier et al. 2003). Beech bark disease may substantially impact wildlife which rely on beech nuts. Another pest that has significantly impacted eastern forests since introduction in the late 19th century is the European gypsy moth. Currently distributed throughout the region, gypsy moth populations fluctuate; during eruptive years, large moth populations cause widespread defoliation of oaks, aspen, and many other trees, including white pine. Successive years of defoliation can kill trees, but annual tree mortality due to gypsy moth seldom exceeds 20% within a region (Morin et al. 2004).

Several other species pose substantial threats if they advance into the region. The Asian longhorned beetle is a large potential threat to maple and other tree species if it invades rural and forested areas from its current documented occurrences in and near New York City. Likewise, the Emerald ash borer is of high concern, though it has not yet been documented in the northeastern US. This insect quickly kills all native species of ash, and could have dramatic impacts if it arrives in this region. Finally, the fungal pathogen that causes sudden oak death has not yet been found in the eastern US, but could have dramatic impacts on oaks and other trees if unwittingly introduced into this region.

Another important taxa currently spreading through northeastern forests are invasive exotic earthworms. Where they occur, earthworms are “keystone” soil fauna which control many aspects of soil structure and nutrient cycling (Hendrix 1995). Native earthworms are rare in northeastern forests, presumably having been removed by glaciation (Gates 1970). Currently, several species of invasive exotic earthworms are spreading through northeastern forests, probably due to introduction by agriculture and fishermen, though the geographical extent of this invasion remains unknown (Hendrix 1995). In northeastern forests recently invaded by exotic earthworms, dramatic impacts have been observed. Most notably, earthworm trophic activity dramatically reduces or eliminates the surface organic horizon or “forest floor” (Alban and Berry 1994) – an important structural feature of temperate forest soils important in nutrient cycling, regeneration and protection from soil erosion (Bormann and Likens 1979). In doing so, earthworms accelerate nutrient cycling, redistribute nutrients vertically among soil horizons and alter availability of key forest nutrients such as phosphorus and nitrogen; these effects are likely to vary depending upon many factors including forest composition, land use history, and species of invading earthworms (Bohlen et al. 2004). Some effects on herbaceous species composition have been observed (Hale et al. 2000).

Several species of invasive exotic terrestrial plants are also substantially impacting northeastern forested ecosystems, by competing with native flora, altering habitat, and altering ecosystem dynamics such as nutrient cycling and hydrology (Mack et al. 2000). A wide variety of invasive exotic plant species are currently present and spreading throughout northeastern forests; a few of the most significant plants are briefly mentioned herein. Norway maple is a hardy and prolific invader of forested habitat. It is very shade tolerant and can dominate regeneration in natural areas near suburban habitat where it has been planted as a landscaping or street tree (Webb et al. 2000). The aggressive shrub, European buckthorn, has escaped from hedgerow and other plantings to become a common species in northeastern forests. A dense understory of exotic Bush honeysuckles smothers many northeastern woodlands, shading out native species and attracting bird dispersers. Likewise, the aggressive European garlic mustard invades a wide variety of open and forested habitats, displacing native spring wildflowers (Welk et al. 2002). Oriental bittersweet is a vine that overshadows forest edges and disturbed woodlands in southern New England and New York, smothering the vegetation beneath.

One additional category of “exotic species” that have a significant presence in some NETN parks is free-ranging and feral cats and dogs. While pet cats and dogs have great to value to humans, the impacts of free-ranging and feral cats and dogs on natural ecosystems are only beginning to be understood. Free-ranging refers to animals which are kept as pets, but which are allowed to roam outdoors freely; feral animals are those that have escaped domesticity to live in the wild, although these animals are sometimes fed by humans. Both cats and dogs are natural predators. A growing body of literature indicates that free-ranging and feral cats are responsible for substantial mortality to small mammals and birds, and perhaps reptiles and amphibians as well (Pearre and Maass 1998, Fitzgerald and Turner 2000, Hall et al. 2000, Lepczyk et al. 2003, Woods et al. 2003). Because free-ranging cats are fed and cared for by humans, they may attain higher population densities than native predators, and thus exert abnormally high predation pressure on natural ecosystems. Moreover, cats are opportunistic predators (Coman and Brunner 1972, Barratt 1997), indicating adequate food supply at home does not prevent cats from predating native wildlife. Home-range of free-ranging cats was estimated using radio-telemetry in one study to vary up to 28 hectares, with the mean range about 8 hectares (Barratt 1997). Dogs are less effective predators than cats, but free-ranging dogs instinctually chase native mammals and other wildlife, which can stress, injure or kills those animals (Sime 1999, Miller et al. 2001). In addition, anecdotal evidence indicates that free-ranging dogs may negatively impact ground vegetation by digging and trampling; this maybe particularly true in areas near trails which are frequented by dog-owners walking their dogs off-leash. Many NETN parks are small, and lie within densely inhabited suburban areas presumably with substantial pet populations; thus several NETN parks may be particularly at risk. Management options for affected parks include enforcement of regulations requiring dogs to be leashed, or prohibiting dogs from sensitive areas; public education campaigns which inform the public about free-ranging pet impacts upon wildlife, and encourage keeping cats indoors; and removal of nearby colonies of feral cats. A variety of wildlife conservation organizations, including the Wildlife Society and the National Audubon Society, advocate some or all of these management strategies.

In addition to exotic species, northeastern forests are impacted by an elevated population of white-tailed deer browsers. In many parts of the northeastern US, deer populations have reached historic high levels due a combination of habitat modification and the extirpation of natural predators (Augustine and DeCalesta 2003). Increased browsing pressure from these large populations can substantially impact tree regeneration, as well as understory and herbaceous species composition. White-tailed deer browsing preference has been shown to inhibit regeneration of hemlock, northern white cedar and some species of oak and birch, and is implicated in the decline of herbaceous diversity in some mixed forests (Rooney and Waller 2003). The impacts of heavy deer browsing can be assessed by monitoring forest regeneration within and outside deer exclosures; deer population sizes may also be monitored from hunting and roadkill records (Halls 1984).

Atmospheric Pollution

Atmospheric pollution, in the form of acid deposition and tropospheric ozone, significantly impacts northeastern forests in complex ways that vary substantially across the landscape. Acidic deposition, derived from emissions from electric utilities, manufacturing, agriculture and other sources, is deposited in precipitation (wet deposition), directly onto vegetation immersed in clouds and fog (occult deposition), and also by direct transfer of particles and gases (dry deposition). Large scale patterns of wet acidic deposition across New England are well characterized - they are most extreme at high elevations and in the southern and western parts of this region which are closest to midwestern emission sources; deposition is slightly lower in central New England and along the Maine coast and lowest in northern and eastern New England (Driscoll et al. 2003). However, substantial additional acidity can result from dry and occult deposition, and these patterns of deposition are not well characterized. Within the NETN, coastal fog at Acadia may deposit substantial acidity (Weathers et al. 1986).

The effects of acidic deposition upon forested ecosystems are complex. Acidic deposition acidifies soil, leaching base cations (e.g., Ca^{2+}, Mg^{2+}, K^{+}) from the soil and increasing availability of aluminum; this deprives vegetation of necessary nutrients, and increases availability of a toxin (Al). Deposition of nitrogen (N) compounds can further alter forested ecosystems - N is a limiting nutrient that has historically been retained within forested systems. As increased N inputs “saturate” forested ecosystems, excess N is leached from the system as nitrate and exacerbates the effects of acidification (Aber et al. 1998). The magnitude of these effects varies spatially across the landscape due to patterns of deposition, species composition, soil buffering capacity, and landuse history (Aber et al. 2004, Lovett et al. 2000). These effects also vary temporally; in particular, spring snowmelt can release substantial acidity accumulated during the winter. The effects of these changes upon northeastern trees have been most well studied for red spruce and sugar maple. A substantial body of evidence indicates that acid deposition causes dieback of red spruce by decreasing cold tolerance due to interference with Ca^{2+} nutrition (Johnson et al. 1984, Shortle et al. 1997, DeHayes et al. 1999). Additional evidence indicates that acid deposition may contribute to sugar maple decline, particularly on marginal sites, due to base cation depletion (Long et al. 1997, Bailey et al. 1999, Horsely et al. 1999). Attempts to control emissions contributing to acidic deposition using federal regulation have yielded some decreases in sulfate deposition and prevented further increases in NO_x emissions, but recovery of affected ecosystems has lagged behind (Driscoll et al. 2003).

The other atmospheric pollutant of chief concern to forest ecosystems, tropospheric ozone is a damaging phytotoxin of significant concern within the northeastern US (US EPA 1996). It is formed by sunlight acting upon nitric oxides and simple hydrocarbons from industrial emissions and motor vehicles. Thus, tropospheric ozone levels vary rapidly in space and time, and are highest on sunny, still days in areas within and downwind of urban centers, industrial facilities and transportation corridors, but elevated background levels of tropospheric ozone occur throughout the northeastern US. Ozone damages cell membranes, which may then reduce rates of photosynthesis and plant growth. However, ozone damage varies in a complex manner depending upon exposure, plant species, genotype, plant age, and plant stress, particularly water stress (Chapelka and Samuelson 1998). For this reason, ozone is typically monitored both directly in air, and indirectly, as injury to indicator species (Coulston et al. 2003).

Climate Change

In addition to these direct stressors, anthropogenic global climate change is both directly and indirectly altering many of the key environmental parameters that control the structure, composition and function of forest ecosystems. While accurate prediction of the effects of the suite of global change stressors upon forested ecosystems is currently beyond our abilities, a large body of research has been assembled which yields some insight into what may occur. Easiest to predict are the direct effects of elevated atmospheric CO_2 concentrations on forest vegetation - elevated CO2 concentrations have been shown to increase photosynthetic rates and tree growth, though this may be a short-term effect (Long et al. 1996, Rey and Jarvis 1998) which is likely to be limited under field conditions by nutrient limitation (Curtis and Wang 1998, Johnson et al. 1998). Enhanced CO_2 should also increase plant water use efficiency, but may reduce tissue N concentrations leaving vegetation more susceptible to herbivory and perhaps altering rates of nutrient cycling (Landolt and Pfenninger 1997, Williams et al. 1998).

It is much harder to predict the effects upon forest ecosystems of changing temperature and precipitation patterns and altered disturbance regimes associated with global change. Current global change predictions suggest that the northeastern US will warm over the next century, particularly during winter and at higher elevations, but it is unclear how patterns of precipitation may change (Mitchell & Johns 1997, Flato et al. 1999). In the short-term, the benefits of enhanced CO_2 may outweigh modest changes in temperature and precipitation causing increased productivity in northeastern forests (Aber et al. 2001). In the mid-term, changing environmental conditions may stress northeastern forests and exacerbate forest decline (Aber et al. 2001). Over the long-term, current predictions suggest that northern hardwood and spruce-fir forests could migrate north out of the US into Canada, and be replaced by oak-hickory and oak-pine forests (Hansen et al. 2001). Species will respond individually to climate change causing the most severe impacts to highly mutualistic species, to poor dispersers, and to populations at the southern extent of their ranges. Current assessments of how global change may alter disturbance regimes within northeastern forested ecosystems are even more speculative, but it seems likely that hurricanes will become more frequent, that disturbances caused by invasive exotic insect pests will become more intense and widespread, and that the geographic extent of ice storms in this region may shrink (Dale et al. 2001).

Thus the effects of some stressors seem predictable based on current scientific understanding, while the effects of others remain quite speculative. The largest issue currently facing scientists and land managers is understanding the cumulative and interactive effects of these varied stressors upon ecosystems. For example, enhanced atmospheric CO_2 concentrations may increase plant water use efficiency, which might reduce foliar ozone exposure (Aber et al. 2001). However, increased disturbance and forest decline associated with global change is expected to increase the spread of invasive exotic species, while migratory responses of native species may be hindered by habitat fragmentation (Hansen et al. 2001). These are just a few examples - the possible interactions between this formidable suite of anthropogenic stressors are numerous, largely unstudied, and in some cases, very much unpredictable.

Wetland Resource Conceptual Model

In this section we present the wetland resource conceptual ecological model as a diagram (Figure 2) accompanied by the following narrative describing our current understanding of wetland system components and their interactions.

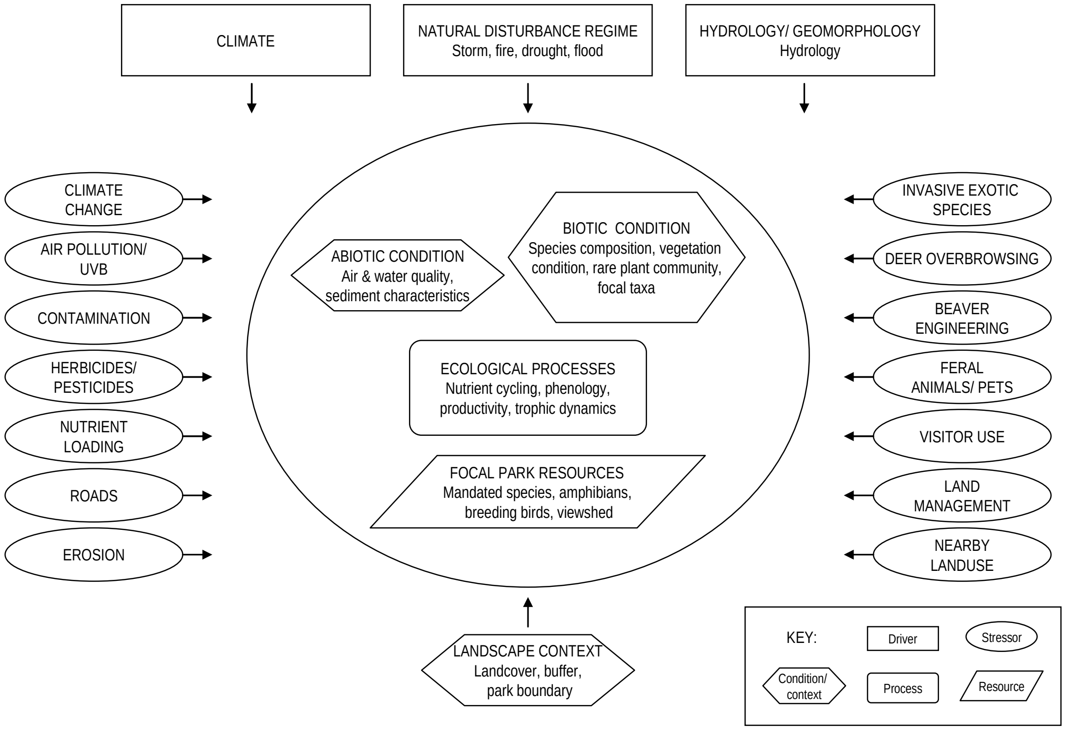

Figure 2. Wetland conceptual model diagram.

<u>Introduction</u>

Wetlands represent a diverse set of ecological communities that occur at the transition between terrestrial and aquatic systems. Defined based on hydrology, physiochemical environment, and biota, wetlands are some of the most productive and diverse ecosystems on earth (Keddy 2000). The physiochemical environment of a wetland is defined as the soils, chemical properties, and processes that interact with the hydrology to influence the biota. These three components form the basis for the development and functioning of wetland ecosystems.

Water is present in all wetlands for some time period but the depth and duration of flooding or *hydrology*, varies substantially among wetland types. Hydrology is the defining physical parameter that separates wetland ecosystems from terrestrial and deep water aquatic systems. Hydrology is thus the single most important factor in the establishment and maintenance of characteristic types of wetlands and wetland processes (Mitsch and Gosselink 2000). Globally, freshwater species and habitats are among the most threatened in the world (Saunders et al. 2002). Freshwater withdrawals have doubled since 1960 and more than half of all freshwater runoff is used by humans (Loh et al. 1998, Saunders et al. 2002). Wetland loss in the United States has been substantial over the past 200 years. Prior to European colonization, wetlands comprised approximately 9% of the continental USA (Dahl 1990), but presently nearly 50% of the wetland area has been converted (NRC 1995). Of the remaining wetlands, only 25% are in government ownership, making the maintenance and conservation of these habitats on federal lands a high priority.

In the northeast United States depressional wetlands and seeps are a priority because of the major function they provide to amphibian breeding (Brinson and Malvarez 2002). These wetlands are most commonly altered or destroyed by urban and suburban development (Brinson and Malvarez 2002), a primary threat to NETN park natural resources. Wetland loss in NETN States has been substantial with an average loss of 38% (± 21%) of the original extent. Connecticut has suffered the most dramatic loss, with 74% of the states wetlands filled or degraded since the 1780's (Mitsch and Gosselink 2000).

Ecological Systems

Wetlands in NETN parks are an important component of park ecological condition and provide valuable habitat to a suite of obligate and facultative wetland flora and fauna. Wetlands are present in all NETN parks (Figures 6, 8, 10, 12, 14, 16, 18, 20, 22, 24) and provide valuable ecological and social services therein. Ecologically, wetlands contribute greatly to biological diversity, provide flood storage, and improve water quality, while socially they provide scenic viewsheds, educational opportunities, and contribute to our natural heritage.

Wetlands in NETN parks are comprised of seven different types of wetland ecological systems (NatureServe 2003) and vernal pools;

1) Laurentian-Acadian Acidic Swamp: These forested wetlands are found in temperate northeastern and north-central U.S., primarily in glaciated regions in the Laurentian-Acadian region. They occur on mineral soils that are nutrient-poor. There may be an organic epipedon, but the substrate is not deep peat. These basin wetlands remain saturated for all or nearly all of the growing season, and may have standing water seasonally. There may be some seepage influence, especially near the periphery. *Acer rubrum, Fraxinus* spp., *Picea rubens* (rarely *Picea mariana*), and *Abies balsamea* are the most typical trees. The herbaceous and shrub layers tend to be fairly species-poor. *Nemopanthus mucronatus* and *Osmunda* spp. are typical shrub and herb species.
2) North-Central Appalachian Acidic Swamp: These swamps are distributed through the Central Appalachians south to Virginia. They are found in basins, or on gently sloping seepage lowlands. The acidic substrate is mineral soil, often with a component of organic muck; if peat is present, it usually forms an organic epipedon over the mineral soil rather than a true peat substrate. *Tsuga canadensis* is usually present and may be dominant. It is often mixed with deciduous wetland trees such as *Acer rubrum* or *Nyssa sylvatica*. *Sphagnum* is an important component of the bryoid layer. Basin swamps tend to be more nutrient-poor and less species-rich than seepage swamps; in some settings, the two occur adjacent to each other with the basin swamp vegetation surrounded by seepage swamp vegetation on its upland periphery.
3) Laurentian-Acadian Floodplain Forest: This system encompasses north-temperate floodplains in the northeastern and north-central U.S. and adjacent Canada at the northern end of the range of silver maple. They occur along medium to large rivers where topography and process have resulted in the development of a complex of upland and wetland temperate alluvial vegetation on generally flat topography. This complex includes floodplain forests, with *Acer saccharinum* characteristic, as well as herbaceous sloughs and shrub wetlands. Most areas are underwater each spring; microtopography determines how long the various habitats are inundated. Associated trees include *Acer rubrum* and *Carpinus caroliniana*, the latter frequent but never abundant. On terraces or in more calcareous areas, *Acer saccharum* or *Quercus rubra* may be locally prominent, with *Betula alleghaniensis* and *Fraxinus* spp. *Salix nigra* is characteristic of the

levees adjacent to the channel. Common shrubs include *Cornus amomum* and *Viburnum* spp. The herb layer in the forested portions often features abundant spring ephemerals, giving way to a fern-dominated understory in many areas by mid-summer. Non-forested wetlands associated with these systems include shrub-dominated and graminoid-herbaceous vegetation.

4) Acadian Maritime Bog: These ombrotrophic acidic peatlands occur along the north Atlantic Coast from downeast Maine east into the Canadian maritimes. When these form in basins, they develop raised plateaus with undulating sedge and dwarf-shrub vegetation. *Trichophorum caespitosum* may form sedge lawns on the raised plateau. The system may also occur as "blanket bogs" over a sloping rocky substrate in extreme maritime settings; here, dwarf-shrubs and *Sphagnum* are the dominant cover. Species characteristic of this maritime setting include *Empetrum nigrum* and *Rubus chamaemorus*. Typical bog heaths such as *Kalmia angustifolia, Kalmia polifolia, Gaylussacia baccata, Ledum groenlandicum*, and *Gaylussacia dumosa* are also present. Morphological characteristics and certain coastal species distinguish these from more inland raised bogs. The distribution is primarily Canadian, and these peatlands are rare in the U.S.

5) Laurentian-Acadian Acidic Basin Fen: This peatland system ranges over a broad geographic area across the glaciated northeast to the Great Lakes and upper Midwest. The fens have developed in open or closed, relatively shallow basins with nutrient-poor and acidic conditions. Many occur in association with larger lakes or streams. The substrate is *Sphagnum*, and vegetation typically includes areas of graminoid dominance and dwarf-shrub dominance. *Chamaedaphne calyculata* is usually present and often dominant. Scattered stunted trees may be present. These fens often develop adjacent to open water. They lack the ribbed or reticulate microtopographical patterning of the patterned fen system.

6) Laurentian-Acadian Alkaline Fen: These fens, distributed across glaciated eastern and central North America, develop in open basins where bedrock or other substrate influence creates circumneutral to calcareous conditions. They are most abundant in areas of limestone bedrock, and widely scattered in areas where calcareous substrates are scarce. The vegetation may be graminoid-dominated, shrub-dominated, or a patchwork of the two; *Dasiphora fruticosa ssp. floribunda* is a common diagnostic shrub. The herbaceous flora is usually species-rich, and includes calciphilic graminoids and forbs. *Sphagnum* dominates the substrate; *Campylium stellatum* is an indicator bryophyte. The edge of the basin may be shallow to deep peat over a sloping substrate, where seepage waters provide nutrients.

7) Laurentian-Acadian Wet Meadow-Shrub Swamp and Marsh: This system encompasses shrub swamps and herbaceous emergent to submergent mineral-soil wetlands of the Northeast and upper Midwest. They are often associated with lakes and ponds, but are also found along streams, where the water level does not fluctuate greatly. The size of occurrences ranges from small pockets to extensive acreages. The emergent wetlands often have a patchwork of shrub and graminoid dominance; typical species include *Alnus incana, Spiraea alba, Myrica gale, Calamagrostis canadensis*, tall *Carex* spp., and *Juncus effusus*. Trees are generally absent and, if present, are scattered. Submergent wetlands include a variety of macrophytes, often with a border of non-persistent emergent vegetation dominated by *Pontederia cordata*. The submergent vegetation zones may be severely impacted by non-native invasive aquatics including *Myriophyllum spicatum* and others.

8) Vernal Pools: Vernal pools are temporary bodies of fresh water inhabited by many species of wildlife, some of which are totally dependent on the pools for their survival (DiMauro and Hunter 2002). Temporary freshwater pools provide critical habitats for breeding populations of amphibians and invertebrates dependent upon fishless environments for successful recruitment (Semlitsch and Bodie 1998). Periodic drying of vernal pools eliminates fish populations and breeding populations of other predators such as bullfrogs (*Rana catesbeiana* Shaw) and green frogs (*Rana clamitans* Latreille). Thus, vernal pools provide a unusual low predation environments for many amphibians. Vernal pools occur throughout North America within both closed canopy and open canopy communities. In northeastern North America, vernal pools are typically found in upland forest and floodplain depression systems that are filled by spring rains, snowmelt, or seasonally raised water tables (Brooks et al. 1998, Brooks and Hayashi 2002). Candidate systems within the Northeast Temperate Network where vernal pools may occur include the following: Laurentian-Acadian Floodplain Forest, Central Appalachian Floodplain, Acadian Lowland Spruce-Fir-

Hardwood Forest, Laurentian-Acadian Northern Hardwood Forest, Laurentian-Acadian Pine-Hemlock-Hardwood Forest, and Appalachian Hemlock-Hardwood Forest.

Ecosystem Drivers

Climate (see introduction of this appendix)

Natural Disturbance Regime

Natural disturbances to wetlands in NETN all influence hydrology and therefore change the abiotic and biotic attributes of the wetland system. Changes to hydrology can occur naturally to wetlands through succession, beaver engineering, sediment transports, severe weather events, and ice scouring. Severe weather events are the most common source of natural disturbance for wetlands in the NETN and determine the extent and duration of floods and droughts. The direct consumption of plants by geese, muskrats, and other herbivores can be common in some wetlands and greatly alters the vegetation composition and structure (Mitsch and Gosselink 2000). Known as “eat outs”, these natural disturbances convert large expanses of wetlands from emergent vegetation to open water (Middleton 1999).

Hydrology/Geomorphology

Hydroperiod (the frequency and duration of soil inundation) defines the hydrology of a specific wetland and largely determines the type of wetland that will develop in a particular setting. Wetland hydroperiod is influenced by basin morphometry, wetland size, connection of the wetland to ground–water resources, and long-term climatic conditions (Larson 1995, Lent et al. 1997, Brooks and Hayashi 2002). Hydroperiod is also closely linked to climatic conditions and annual variation in hydroperiod is thought to be an expression of the annual variation in weather patterns, specifically precipitation (Winter et al. 2001). Moreover, hydroperiod is the most important physical factor driving the composition and diversity of the wetland floral and faunal communities and wetland productivity (Wiggins et al. 1980, Semlitch et al. 1996, Schneider 1999, Mitsch and Gosselink 2000, Brooks 2004). Therefore, monitoring wetland hydroperiod not only provides detailed information about wetland condition, structure, and function but also can be used as a corollary to better understand the ecological effects of changing weather patterns.

Hydroperiod is likely the most important factor in determining wetland structure, function, and condition. (Weyrauch and Grubb 2004) found hydroperiod to be the most important variable in predicting amphibian species richness and showed a positive relationship between hydroperiod and species richness. Snodgrass et al. (2000) recommended that an array of small wetlands with variable hydroperiods be conserved in order to maintain biological diversity at the landscape scale. Wetlands of shorter hydroperiods and smaller size are likely to support species not found in permanent wetlands because of the absence of fish predation in ephemeral wetland (Semlitsch and Bodie 1998, Gibbs 2000, Snodgrass et al. 2000). Alterations to hydroperiod that increase flooding can reduce the extent of emergent vegetation, alter the benthic community, and decrease water clarity. Hydrologic alterations that increase the duration of water level drawdown periods reduce wetland size, limit amphibian breeding, and can increase the probability of invasion by exotic plant species.

The geomorphic setting of the wetland is important in determining wetland type and the dominant sources of water a wetland receives (Brinson et al. 1998). Wetlands of different geomorphic settings usually receive different sources of water, have different hydroperiods and therefore, different species compositions (Brinson et al. 1998, Mitsch and Gosselink 2000). Wetlands with similar geomorphic settings tend to be subjected to the same anthropogenic stressors (Brinson and Malvarez 2002). Depressional wetland hydrology is tightly correlated with groundwater levels making this wetland type subject to complete drying during periods of groundwater withdrawal. Periods of complete drying can eliminate the role of depressional wetlands in maintaining wetland faunal diversity, especially amphibians (Semlitsch and Bodie 1998).

Ecosystem Condition

Abiotc Condition

There is a wide range of chemical conditions in wetland soils and water (Mitsch and Gosselink 2000). The chemistry of freshwater marshes differs substantially from the chemistry of ombrotrophic bogs, with differences related to the magnitude of nutrient inputs and the relative importance of ground water and surface water inflow (Mitsch and Gosselink 2000). Inflowing water to freshwater wetlands tends to have higher amounts of dissolved minerals, including nutrients, compared to bogs that are fed by precipitation only. The pH of most freshwater marshes is generally circumneutral to slightly basic whereas the pH of bogs generally decreases as the organic content increases, creating a gradient of soil acidity among different wetland types (Mitsch and Gosselink 2000).

Nutrient concentrations in wetland systems are greatly influenced by flooding events, connection with groundwater, nutrient uptake by plants, and substrate or parent material (Mitsch and Gosselink 2000). Flooding in wetland systems is controlled by seasonal changes in precipitation, runoff, and evapotranspiration, and in the northeast occurs most frequently during the spring and fall.

Amphibians are especially sensitive to wetland water chemistry and ion concentration is particularly important for amphibian development (Cook 1983, Freda and Dunson 1986, Portnoy 1990, Turtle 2000). Low pH can be especially detrimental to developing amphibian embryos and Portnoy (1990) observed complete mortality of spotted salamander (*Ambystoma maculatum*) embryos in vernal pools having a pH of 4 or lower. Turtle (2000) found that de-icing salts heavily contaminate roadside wetlands and reduced spotted salamander survivorship.

Biotic Condition

Algae are an important component of wetland systems and are often a more important source of energy than vascular plants (Neill and Cornwell 1992, Peterson et al. 1993, Adamus et al. 2001). Algae are commonly grouped based on where they occur in the vertical strata. Phytoplankton are algal species suspended in the water column, metophyton are unattached and floating, benthic algae are attached to the substratum, and epiphytic algae are attached to plants (Adamus et al. 2001). Algal production is often limited by phosphorous and nitrogen and most algal groups are sensitive to changes in the concentrations of these macronutrients (Bothwell 1989) making them potential indicators of eutrophication and nutrient enrichment.

Vascular plants, or macrophytes, are increasingly being used as indicators of wetland condition (Adamus et al. 2001). Macrophytes are commonly used to delineate wetland boundaries and to classify wetland types. Common plant species in northeast wetlands include:red maple (*Acer rubrum*), silver maple (*Acer saccharum*), green ash (*Fraxinus pennsylvanica*), buttonbush (*Cephalanthus occidentalis*), meadow-sweet (*Spiraea alba*), speckled alder (*Alnus incana*), willow (*Salix spp.*), common cattail (*Typha latifolia*), pickerelweed (*Pontederia cordata*), broad-leaved arrowhead (*Sagittaria latifolia*), and sphagnum mosses (*Sphagnum spp.*).

Wetland invertebrates are important trophic links between plants and their detritus and animals and fish (Mitsch and Gosselink 2000). Many groups of insects serve important roles in wetland nutrient cycling by shredding plant material to increase availability to bacteria (Adamus et al. 2001). Invertebrate fauna are increasingly being used as indicators of wetland condition (Adamus et al. 2001). Some invertebrate species, such as fairy shrimp (*Eubranchipus* spp.), are also entirely dependent upon vernal pool habitat and many species act as important predators and prey in wetland ecosystems (King et al. 1996).

Amphibians and reptiles are the dominant vertebrate groups in many freshwater systems of NETN parks. Common species include the american toad (*Bufo americanus*), green frog (*Rana clamitans*), american bullfrog (*Rana catesbiena*), gray treefrog (*Hyla versicolor*), pickeral frog (*Rana palustris*), spring peeper (*Pseudacris crucifer*), eastern newt (*Notophthalmus viridescens*), painted turtle (*Chrysemys picta*), Blanding's Turtle (*Emydoidea blandingii*), and snapping turtle (*Chelydra serpentina*). Some species like, wood frog (*Rana sylvatica*), the eastern spadefoot toad (*Scaphiopus h. holbrooki*), and the four species of mole salamander (*Ambystoma* spp.) have evolved breeding strategies intolerant of fish predation and are considered vernal pool obligate breeders. The lack of fish populations is essential to the breeding success of these species. Vernal pools are a high conservation priority in the northeast due to the loss of vernal pools and general lack of regulatory protection for these ephemeral habitats.

Other dominant wetland faunal groups include mammals and birds. Beaver (*Castor canadensis*) and muskrat (*Ondatra zibethicus*) are common in NETN wetlands, both of which can cause major changes in marsh vegetation structure and composition. Common wetland avi-fauna include least bittern (*Ixobrychus exilis*), american bittern (*Botaurus lentiginosus*), great blue heron (*Ardea herodias*), black-crowned night heron (*Nycticorax nycticorax*), wood duck (*Aix sponsa*), black duck (*Anas rubripes*), virginia rail (*Rallus limicola*), Sora (*Porzana carolina*), marsh wren (*Cistothorus palustris*), northern waterthrush (*Seiurus noveboracensis*), and red-winged blackbird (*Agelaius phoeniceus*).

Ecological Processes

Nutrient Cycling

A major feature that separates wetland from terrestrial systems is the anaerobic nature of wetland soils (Morris 1991). The absence of oxygen in wetland soils slows the decomposition of organic material compared to terrestrial systems. Wetlands, because of the gradients in available oxygen, maintain the widest range of oxidation-reduction reactions of any ecosystem type (Keddy 2000). This effectively allows wetlands to function as transformers of nutrients and metals where elements are converted among an array of chemical states (Mitsch and Gosselink 2000). Wetland nutrient cycling is dominated by the detritus food web where bacteria and invertebrates are a key component in nutrient cycling. Most nitrogen is stored in these organic sediments. Nitrogen cycling within a wetland is controlled by the temperature, pH, and the amount of available oxygen (Keddy 2000). Both nitrogen and phosphorous are limiting to wetland productivity and are often interdependent (Mitsch and Gosselink 2000).

Productivity

Wetland primary productivity estimates are high and range from 500-6000 g $m^{-2}yr^{-1}$ (Mitsch and Gosselink 2000). Productivity varies among wetland types, primarily dependent on wetland hydrology. Wetlands with flowthrough hydrology (e.g., fens) tend to have higher primary productivity than stagnant, ombrotrophic systems (e.g. bogs, Mitsch and Gosselink 2000). Hydrology, the main pathway through which nutrients are transported into and through wetlands, greatly influences wetland productivity.

Wetlands are pulsed ecosystems where changes in water levels influence the flow of nutrients and oxygen and therefore, productivity. Wetland productivity is also related to the efficient functioning of both the grazing and detritus food chains. In bogs, where there is little or no pulsing, organic matter accumulates to form peat rather than being broken down and released into the system. Variation in productivity can be primarily attributed to summer temperatures, with a positive relationship between temperature and productivity (Gorham 1974) and genetic differences among plant species in photosynthetic efficiency (Kvet and Husak 1978).

Focal Park Resources

Amphibians and Reptiles

Amphibians occur in all NETN parks and are the organisms most sensitive to changes in water quality, wetland condition, and wetland landscape context. Amphibians are experiencing species extinctions and population declines globally with causes ranging from direct habitat destruction (Blaustein and Wake 1990, Fellers and Drost 1993), changing climate (Rohr and Madison 2003), disease (Blaustein et al. 1994), contaminants (Boyer 1993, Beattie and Tyler-Jones 1992), and introduced species (Hayes and Jennings 1986). The extreme sensitivity of amphibians to environmental stressors, their ubiquitous distribution in the northeast, and their importance to the public, make this group an important focal resource to be included in a long-term monitoring program.

Declines and extinctions in amphibian species are not limited to heavily developed or degraded areas. Two species – the Northern Dusky Salamander (*Desmognathus fuscus*) and the Northern Leopard Frog (*Rana pipiens*) – were once present in Acadia NP (Manville 1939, Coman 1987), but recent inventory work did not detect these species (Behler et al. 2004). The potential loss of two species from Acadia NP is troubling and emphasizes the need for monitoring amphibian population status, not only as an indicator of wetland condition, but as early warning of species population declines.

Most amphibians and many reptiles require aquatic habitats during some stage of their annual cycle and are therefore especially vulnerable to wetland alteration and/or contamination (Dodd and Cade 1998, Stebbins and Cohen 1995, Lannoo 1998, Olson and Leonard 1997). Amphibians and reptiles, because of their limited dispersal ability and migration patterns, are especially sensitive to the landscape matrix surrounding wetlands (Gibbs 1998a, b).

Landscape Context

Habitat fragmentation and buffer loss are major anthropogenic stressors to surrounding wetland habitats. Freshwater systems are affected by any land-use activities occurring upstream. Land-use practices that alter land cover types to reduce native vegetation can affect fresh water systems by modifying nutrient loads, sediment accretion, water temperature, and contaminant inputs (Saunders et al. 2002).

Wetland density on the landscape is often cited as an important explanatory variable in population and community level studies of species persistence and distribution (Fahrig and Merriam 1985, Kotliar and Wiens 1990). High wetland density on the landscape may reduce the risk of local species extirpation by minimizing costs associated with dispersal and maintaining hydrologic regimes (Morris 1992, Mitsch and Gosselink 2000). Landscape orientation of wetlands, especially small isolated wetlands, is a critical determinant of obligate species population viability (Gibbs 1993, Guerry and Hunter 2002). Wetlands tend to be spatially aggregated and hydrologically linked (Brooks et al. 1998). Dispersal opportunities among wetlands are needed to maintain viable populations of organisms dependent on wetland habitats. In this context, metapopulation models may serve as a basis for understanding amphibian dispersal and colonization behavior. However, quantitative information needed for effective population modeling is lacking (Brooks et al. 1998).

Wetland patch size has also been shown to be an important metric in determining wetland condition and many studies have shown that patch size is fundamentally important in maintaining biodiversity (Fahrig and Merriam 1985, Robinson et al. 1992).

Amphibians and reptiles are especially sensitive to the matrix of habitats surrounding a wetland because they spend the majority of their lives foraging, resting, and hibernating in the surrounding terrestrial habitat (Semlitsch 1998). Upland habitats immediately surrounding wetlands serve as important dispersal corridors and are also used as foraging and aestivation areas for many amphibian species (Semlitsch 1998). Weyrauch and Grubb (2004) found woodlot characteristics surrounding the wetland to be the most important variables predicting caudates and salamander species richness. Semlitsch (1998) monitored terrestrial migrations for six Ambystomid salamander species and concluded buffer areas 164 m from wetland edges were needed to encompass 95% of population forays. Total forested area around the wetland also seems to be an important landscape component in the maintenance of wetland fauna. Guerry and Hunter (2002) found that wood frogs, green frogs, eastern newts, spotted salamanders, and salamanders of the blue-spotted/Jefferson's complex (*Ambystoma laterale*/*A. jeffersonianum*) were more likely to occupy ponds in more forested areas than areas with low forest cover.

Roads are among the most widespread forms of habitat modification on the landscape and can have profound effects on wetland communities Trombulak and Frissell 2000, DiMauro and Hunter, Jr. 2002, Gibbs and Shriver 2002, Forman et al. 2003). Road construction has been implicated in the significant loss of wetland biodiversity at both local and regional scales for birds, herptiles, and vascular plants (Findley and Houlahan 1997). Fragmentation resulting from road construction and the associated traffic intensity can act as a barrier to amphibian movement and reduce amphibian abundance (Fahrig et al. 1995, Gibbs 1998a; DeMaynadier and Hunter 2000). The combined effects of ionic inputs, edge effects (DeMaynadier and Hunter 1998), and adult mortality make roads an important landscape component to monitor when estimating wetland condition. (Findley and Bourdages 2000) documented lag times associated with the negative response of species richness to road construction and suggested that the affects of road construction may not be detectable for decades.

Stressors

Atmospheric Deposition

Air pollution is deposited in wetlands and can affect water quality in these systems. The rise in anthropogenic releases of nitrogen oxides increases the deposition of these compounds into wetland systems and can alter wetland structure and function (Morris 1991). Similar to nitrogen, mercury is a naturally occurring element, but studies show that human activities have more than tripled its concentration in the environment, which can cause negative impacts to wetland systems (Mason et al. 1994).

The deposition of nitrogen is the major stressor to wetland systems caused by atmospheric deposition. Bodies of water receiving elevated amounts of nitrogen compounds often show signs of water quality degradation. Atmospheric deposition of nitrogen occurs as wet (in precipitation) and dry (sorption of nitrogen gasses by wet surfaces) deposition and through the capture of fog or cloud droplets by vegetation (occult, Morris 1991). Atmospheric deposition of nitrogen compounds can lead to eutrophication, or harmful increases in the growth of algae within wetland systems. Nitrogen pollution and the resulting eutrophication of wetlands, alters species composition of both flora and fauna and in some cases, nitrogen pollution can contribute to the acidification of water bodies.

Acidification is also common in water bodies in the eastern United States where weather patterns deposit acids made from air pollutants generated in the Midwest and points further west. The effects of acid deposition depend greatly upon characteristics of the water body in which they are deposited. Aquatic organisms in acidified waters often suffer from calcium deficiency, which weakens bones and exoskeletons and can cause eggs to be weak or brittle. It also affects the permeability of fish membranes and particularly, the ability of gills to take in oxygen from water. Increasing amounts of acid in a wetland change the mobility of certain trace metals like aluminum, cadmium, manganese, iron, arsenic, and mercury. Acid deposition has lowered the pH, decreased the acid-neutralizing capacity (ANC), and increased the aluminum concentrations causing a decline in aquatic species diversity and abundance in the northeast (Driscoll et al. 2001). Many amphibian species are susceptible to increasing pH, especially those breeding in temporary wetlands or vernal pools. Permanent wetlands may have a natural buffer capacity to neutralize acidification but ephemeral wetlands are created by snow melt or spring runoff which tends to concentrate acid and lower pH (Hunter et al. 1992). Pough (1976) showed that for spotted salamanders, hatching rates declined and deformities increased rapidly as pH dropped below 7. Algae are affected by acidification as a result of direct toxicity and changes in competition with organisms less sensitive to rising pH (Adamus et al. 2001). Either extreme of acidity can diminish species richness of algal communities.

Hydrologic Alterations

Beaver engineering is one of the most pervasive hydrologic alterations to NETN park wetlands. Water diversions of any kind can be viewed as potential agents of both positive and negative change to wetlands. Beaver can affect almost any wetland type but are especially common along streams and ponds where they build dams. Dam construction typically kills all woody vegetation, reduces the water velocity, and drastically changes plant species composition and structure (Thompson and Sorenson 2000). Beaver alteration of wetlands occurs in decadal cycles with an initial period of flooding after dam creation and impoundment followed by abandonment after the beaver deplete the food source. Thus, beavers both destroy through flooding unusual vegetation of bogs and fens, for example, but conversely create many highly productive wetlands along streams formerly dominated by upland vegetation.

Climate Change

A growing body of evidence suggests that human activities have accelerated the concentration of greenhouse gases in the atmosphere that cause rapid climate changes (IPCC 2001). Atmospheric models predict the average surface temperature to rise from 1.4 to 5.8ºC by 2100 (IPCC 2001). The climate of the northeastern United Sates is projected to become warmer and wetter over the next 100 yrs (New England Regional Assessment Group 2001), changes that will likely affect the structure, function, and distribution of wetlands. Both annual and seasonal minimum temperatures are predicated to increase at a greater rate than maximum temperatures (Brooks 2004). These projected increases in temperature would also increase the rate of evapotranspiration which in turn would alter wetland hydrology. Hydrologic alterations that reduced the flooding period would have the most negative impacts on ephemeral wetland or vernal pools (Brooks 2004). Changes in wetland water temperature due to rapidly changing climate are also predicted to alter the sex ratios of turtle populations because of their temperature dependent sex determination (Root and Schneider 2002). Wetland herpetofauna may be especially sensitive to changing climate caused by the synergistic effects of habitat fragmentation and the increased need for dispersal caused by a reduction in habitat quality. Increases in the rate of temperature change to wetland habits may force many individuals to disperse more frequently. As landscape matrices have become more hostile to dispersing wetland herptiles the increase dispersal may reduce populations and further bias sex ratios (Gibbs and Shriver 2002, Steen and Gibbs 2004).

Climate change, specifically increases in temperature, have already been shown to change the breeding and dispersal phenology of many species (Schneider and Root 2002). Climate change is anticipated to have a pronounced effect on freshwater ecosystems, especially those in northern latitudes (IPCC 2002). The combined effects of changes in temperature and precipitation are likely to severely alter wetland hydrology and water quality, thus jeopardizing the flora and fauna dependent on these systems. In the northeast, several frogs have advanced their first calling dates by 10-13 days since the early 1900's (Gibbs and Breisch 2001). Because amphibians are especially sensitive to temperature they can be valuable indicators of the biotic response to climate change in wetland systems.

Contamination

Contamination for this model is defined as the increase in concentration, availability, and/or toxicity of metals and synthetic substances (Adamus et al. 2001). Wetland contamination is typically associated with runoff from agricultural areas, residential and urban areas, waste water treatment facilities, and atmospheric deposition.

Heavy Metals-Heavy metals such as mercury, lead, zinc, and cadmium can be directly toxic to wetland fauna (Adamus et al. 2001). Mercury is especially problematic in the northeast where deposition is high. When mercury becomes deposited within a water body, microorganisms can transform it into a very toxic substance known as *methyl mercury*. Methyl mercury can accumulate in the tissues of fish to concentrations much higher than in the surrounding water. *Methyl mercury* tends to remain dissolved in water and does not travel very far in the atmosphere.

Combustion Emissions-Dioxins and furans are families of chemicals that are present in *combustion emissions* and are known to be highly toxic to wildlife. The most toxic dioxin compound is 2,3,7,8-tetrachlorodibenzo-p-dioxin or TCDD. In animal populations, TCDD has been shown to disrupt the endocrine system, weaken immune systems, and cause reproductive damage to wildlife populations. Similarly, the most toxic furan is a compound known as 2,3,7,8-tetrachlorodibenzofuran or TCDF. In animals, furans can cause serious damage to the stomach, liver, kidneys, and immune system. Both families of compounds are persistent in the environment and can concentrate in the tissue of fish and other animals.

Polycyclic aromatic hydrocarbons (PAHs) are a complex mix of compounds that occur in soot and exhaust from automobiles and the incineration of many different materials.

Polychlorinated biphenyls (PCBs) are extremely stable and can concentrate in the tissues of aquatic animals. Concentrations of PCBs in the tissue of some animals can reach literally hundreds of thousands of times greater than the surrounding water. PCBs can cause bronchitis, irritation of the gastrointestinal tract, nervous system impairment, fertility problems, and changes in liver function. They have been shown to cause cancer in lab animals, and are a suspected human carcinogen.

Invasive Species

Invasion of native habitats by nonindigenous species or by native species whose densities are becoming unnaturally inflated (e.g., white-tailed deer) is presently recognized as second only to direct habitat loss and fragmentation as a threat to biodiversity (Walker and Steffen 1997). Pimentel et al. (2000) estimated that invasive species cost the United States $138 billion annually making the reduction of these species a shared priority of many agencies and organizations in the United States (National Invasive Species Council 2001). Once viable populations of invasive plants become established in novel habitats, they inflict a suite of ecological damage to native species including loss of habitat, loss of biodiversity, decreased nutrition for herbivores, competitive dominance, overgrowth, struggling, and shading, resource depletion, alteration of biomass, energy cycling, productivity, and nutrient cycling (Dukes and Mooney 1999). Invasive plant species can also affect hydrologic function and balance, making water scarce for native species (Enright 2000).

Wetland invasive plant species in NETN parks presently include, but are not limited to purple loosestrife (*Lythrum salicaria*), japanese knotweed (*Polygonum cuspidatum*), water chestnut (*Trapa natans*), flowering rush (*Butomus umbellatus*), yellow iris (*Iris pseudacorus*) and phragmites (*Phragmites australis*). These species have been detected in most parks and cause a substantial management effort to control and reduce wetland condition. Invasive plants significantly alter species composition and diversity and often form monotypic stands.

Nutrient Enrichment

Sources of nutrients, especially nitrogen and phosphorous, enter wetlands via surface water, groundwater, and the atmosphere (Brinson and Malvarez 2002) and can dramatically change the composition of both the floral and faunal communities (Bedford et al. 1999). Nutrient enrichment also increases the risk of invasive species establishment in many wetlands, a primary threat to NETN wetland and terrestrial resources. Increases in nitrogen and phosphorous in wetlands causes eutrophication, often at concentrations that exceed natural levels. The dominant source of nutrient inputs into wetland systems comes from agricultural and residential runoff. The most obvious response of wetland systems to nutrient enrichment are harmful algal blooms where algal biomass increases rapidly (Humphrey and Stevenson 1992). Eutrophication can also lead to simplification of algal communities expressed by a decrease in species richness, diversity, and evenness (Steinman and McIntire 1990). Changes in nutrient concentrations can alter macroinvertebrate populations which in turn can change the trophic dynamics between those that consume algae and those that consume vascular plants (Adamus et al. 2001).

Excessive nutrients can affect the wetland plant communities by: 1) shifting species composition from dominance of species that uptake nutrients slowly to those that exploit rapid pulses of nutrients, 2) triggering algal blooms that can shade out submersed herbaceous plants, and 3) causing dead plant material to accumulate faster than it can decompose (Adamus et al. 2001). Wetlands exposed to long-term nutrient enrichment tend to have lower plant species richness than reference wetlands. Bogs and nutrient poor wetlands are most sensitive to the negative effects of nutrient enrichment.

Soil Erosion/Sedimentation

Sedimentation is a naturally occurring process in wetland systems, but accelerated rates can have negative effects on wetland condition. Sedimentation is regarded as one of the major threats to fresh-water aquatic systems, primarily due to the effects of burial (Richter et al. 1997). Increased rates of sedimentation can affect wetlands by adding sediment-born pollutants, burial of vegetation and seed banks (Neely and Baker 1989, Childers and Gosselink 1990), and change the water depth and hydroperiod. Burial can smother aquatic invertebrates and fish eggs, and reduce oxygen availability by stimulating plant growth through nutrient addition (Keddy 2000).

Aquatic Resource Conceptual Model

In this section we present the aquatic resource conceptual ecological model as a diagram (Figure 3) accompanied by the following narrative describing our current understanding of aquatic system components and their interactions.

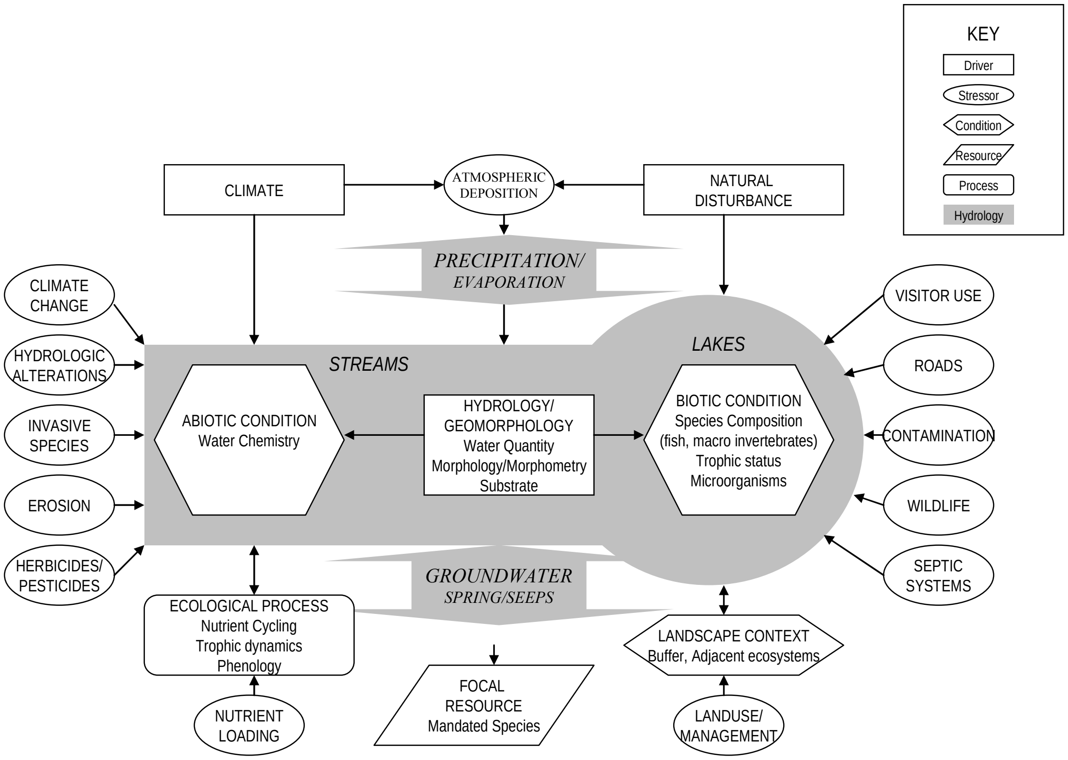

Figure 3. Aquatic resources conceptual model diagram.

Ecological systems

Freshwater aquatic ecosystems within NETN parks consist of lakes, ponds, streams, groundwater, and springs/seeps (Figures 6, 8, 10, 12, 14, 16, 18, 20, 22, 24). These ecosystems are the result of glacial ice sheets during the past 2.5 million years. Ice sheets deepened valleys, and transported and deposited vast quantities of sediment upon scoured bedrock as glacial drift (Randall 2001). Currently, the topographic landscape varies from rolling to mountainous upon mostly acidic bedrock and glacial till.

Lakes and Ponds

Lakes and ponds are found where the water table is at or above land surface, and are generally areas of ground-water discharge. The majority of the lake basins in the northeast were caused by unequal thickness of -drift deposited in preglacial valleys; the thicker masses constituting dams. Thousands of such basins held lakes when the ice vanished and have since became swamps or meadows (Fenneman 1938).

Nine NETN parks contain ponds smaller than 15 acres, many of which are man-made impoundments that pre-date the establishment of the parks. ACAD is the only park in which numerous lakes greater than 15 acres are a dominant part of the landscape. Lakes and ponds within NETN parks vary in type, size and trophic status.

Streams and Rivers

Streams and rivers within NETN parks vary from first order headwater streams to tidal rivers. Drainage patterns of northeastern streams were altered by the last glaciation. As drift was deposited in varying thicknesses, dams were created and channels blocked. Streams followed a new course based on the slope of the drift surface. After a stream cut through the drift, it often crossed ridges or ledges of hard rock and developed falls and rapids, eventually carving gorges disproportionate to the changes in relief (Fenneman 1938). Several of the parks border large rivers such as the Hudson River and the Connecticut River, and are occasionally impacted by these larger river systems during times of high water.

Information about groundwater resources and the properties and extent of aquifers making up this resource varies greatly in the northeast. The stratified drift deposited by -glaciers includes a nonsorted, nonstratified mixture of grain sizes from clay and silt to large boulders. These sand and gravel deposits comprise the most productive aquifers in the northeast. These aquifers vary greatly in grain size, water-transmitting properties, and saturated thickness of the stratified drift (Randall 2001). Groundwater is also influenced by the type of bedrock underlying the stratified drift. The igneous and metamorphic bedrock that underlies much of the northeast has a hydraulic conductivity that is much lower than that of stratified drift (Randall et al. 1988). Ground-water/surface-water interactions occur in both directions and occur throughout the parks.

Springs occur where the water table intercepts the ground surface and discharge is sufficient to flow most of the time. If no flow is evident, the resulting wet areas are called seeps. Discharge of the spring is determined by the permeability and recharge of the aquifer, and thus can indicate the location and extent of an aquifer. Springs can be found at the toe of hillslopes, along depressions such as stream channels, and where the ground surface intercepts an aquifer covered by an aquiclude – a geologic barrier to groundwater flow (Brooks et al. 1991). NETN parks vary in the presence and quantity of springs and seeps.

Ecosystem Drivers

Climate

Climate is one of the main drivers affecting ecological properties and processes in aquatic ecosystems. The temperate climate of the northeastern U.S. is described earlier; briefly, it is characterized by changeable weather, wide ranges in diurnal and annual temperatures, distinct seasonal trends, and precipitation distributed evenly throughout the year. Disturbance regimes are another major driver affecting aquatic ecosystems. Floods and droughts are the primary disturbances that affect aquatic ecosystems in NETN parks. Floods can occur during any season in the northeast, but are most widespread in the spring when large frontal systems bring steady rain which falls on frozen or saturated ground. In the summer and fall, thunderstorms and hurricanes can cause local flooding (Maloney and Bartlett, 1991). Floods are natural recurring events that can cause major morphological shifts in river systems, and cause widespread erosion and sedimentation.

Droughts are more difficult to define and quantify than floods, but are also natural recurring events in the northeast. Hydrologic drought can be defined as reduced streamflows, declining ground-water levels and/or reductions in lake or reservoir levels (American Meteorological Society, 1997). They can be widespread across the entire northeast, or affect only parts of the region. Droughts affect all aspects of water quantity and water quality, and can result in water use conflicts.

Long-term streamflow records show both floods and droughts and thus reflect the natural range of variability in aquatic ecosystems. Short-term records may be skewed depending upon the period of record.

Aspects of the hydrology/geomorphology which characterize NETN parks also drive the ecological properties and processes operating in the park aquatic systems, and are described above under ecological systems.

Ecosystem condition

Understanding and tracking ecosystem condition is an integral part of the Inventory and Monitoring program and is directly addressed in two of the program goals. The condition of the ecosystem affects the status of the aquatic habitat and the condition of individual species. The ecological condition of freshwater resources includes abiotic attributes such as water chemistry as well as biotic attributes such as trophic status, species composition of selected taxa, and microorganisms.

The abiotic condition of water chemistry is widely applicable and is critical for interpreting the biotic condition and status of ecological processes of a resource. Measures such as water temperature, dissolved oxygen and pH define the habitat in which various flora and fauna can survive. Water chemistry affects the trophic status of an ecosystem, the metabolism of aquatic species and the bioavailability of contaminants.

Vital signs that reflect biotic condition, such as composition and abundance of fish, macroinvertebrates and zooplankton, are highly relevant indicators of ecosystem condition that integrate the state of physical, chemical, and biological attributes of the environment. Fish communities integrate their physical, chemical, and biological environment through multiple years while macroinvertebrates do so over a single year. Zooplankton community composition and abundance is indicative of the trophic status of lakes, and reflects primary and secondary production (Porter 1977).

Ecological process

Nutrient Cycling and Productivity

Nutrient cycling, or the movement of nutrients through the water, plants, animals and sediments of freshwater ecosystems, is linked to the productivity and function of these ecosystems. The trophic status of a waterbody is also a measure of its productivity, or the rate at which organic matter is produced. The invertebrates, algae, bryophytes, vascular plants, and bacteria of freshwater systems, which are responsible for much of the work of nutrient cycling, are adapted to the specific sediment and organic matter conditions of their environment and are thus sensitive to changes in the type, size, or frequency of sediment inputs. Understanding nutrient cycling and productivity in NETN aquatic systems may provide links between ecosystem condition, ecosystem function, and stressors such as nonpoint source pollution and land use.

Phenology

Phenology, or the response of living organisms to seasonal and climactic changes in the environment, is a process that helps define the condition of the ecosystem. The combined effects of climate change and other stressors have the potential to substantially alter hydrological and biogeochemical processes, and thus, the floral and faunal communities of freshwater ecosystems of the New England/Mid Atlantic region (Moore et al. 1997). A century of data on lake ice-out dates within the northeastern US shows an advance in spring ice-melt (Hodgkins et al. 2002). Similarly, a shorter record shows an advance in the timing of spring peak river flow (Hodgkins et al. 2003). These trends could have implications for the ecological integrity of lakes and streams in that they could potentially lead to lower oxygen levels, eutrophication, and shifts in floral and faunal communities. Further research is needed to better understand relations between the documented trends in freshwater systems, and the ecological integrity of these systems.

Landscape context

The relationship between freshwater aquatic ecosystems and the surrounding landscape contributes to the condition of these ecosystems. The dimensions of a river channel reflect the interplay between the ability of water to erode the land surface and the ability of the land surface to resist erosion. Landuses such as farming, forestry, development, and water management can all affect the magnitude and frequency of streamflow and thus a river's ability to erode the land. When streams are constrained from meandering by urban alterations, hydraulic instability can cause increased deposition, erosion, slumping, over- widening or the abandonment of existing channels for new ones (Dunne and Leopold 1978, Rosgen 1996). As waterbody buffers expand or contract, sources and amounts of nonpoint source pollution and runoff to the waterbody can also change. Barriers between waterbodies, such as impoundments, can inhibit the movement of species and thus affect the floral and faunal composition of a waterbody. Landscape context is linked to and reflects changing landuse, which is further discussed in the section below regarding stressors.

Stressors

Stressors to freshwater ecosystems in NETN parks include physical, chemical, or biological perturbations acting both inside and outside park boundaries. Many parks are not self-contained watersheds, but are at lower points in a watershed, and can be greatly influenced by alterations to the rivers, streams, or lakes that occur upstream from the parks. Furthermore, some areas that are now protected within parks were substantially altered prior to protection as a national park unit. Thus threats to freshwater aquatic resources in NETN parks cannot be evaluated without examining current and historic landuse in the region. As the landscape of the region has been transformed from agricultural to urban over the past 150 years, the increased use of automobiles and shifts in predominant industry from paper and textile mills to high-tech industry have had substantial impacts on the region's freshwater ecosystems Stressors currently acting on freshwater resources within NETN parks include climate change, atmospheric pollution, contaminants/toxics, nutrient loading, hydrologic alterations, erosion, herbicides/ pesticides from both agricultural and residential use, roads, landuse and land management, visitor use, exotic invasive species, and beaver populations.

Climate Change

Climate Change has the potential to affect the abiotic and biotic condition of freshwater resources across the region. Several geophysical and biological studies indicate that spring is coming earlier in New England. The annual date of the last hard spring freeze became significantly earlier from 1961 to 1990 (Cooter and Leduc 1995) and lilac blooms dates at 4 stations became significantly earlier from 1959 to 1993 (Schwartz and Reiter 2000). The impacts of climate change on hydrology in the northeast are just beginning to be understood. Much of the significant change toward earlier lake ice-out dates in New England since the 1800's occurred from 1968 to 2000 (Hodgkins et al. 2003). All of 11 studied rivers in New England had significantly earlier winter/spring high flows from earlier snowmelt, with most of the change occurring in the last 30 years (Hodgkins et al. 2003). Furthermore, snow density on or near March 1 has significantly increased in coastal Maine over the last 60 years, indicating earlier spring melting? (Dudley and Hodgkins 2002).

Atmospheric Pollution

Atmospheric Pollution in the form of acid deposition is the largest source of nitrogen to streams in the northeast. Measures of atmospheric deposition are critical for understanding water chemistry and stress (Likens and Bormann 1974). Fifty percent of total nitrogen entering New England rivers and streams in 1992-1993 was estimated to come from atmospheric deposition originating both inside and outside the region (Moore et al. 2004) Atmospheric deposition is particularly problematic in NETN parks for the surface water bodies with low ANC. ANC is a key indicator of recovery, determining the capacity of lakes and streams to buffer acidic inputs and prevent further acidification. The relationship between atmospheric pollution and the acid base status of surface waters is complex and nonlinear. The complexities of ecosystem response can be due to confounding factors such as climate change affecting water chemistry, and the natural organic acidity of surface waters. In order for the relationship to be better understood, long term monitoring of atmospheric deposition as well as of the water chemistry of freshwater ecosystems is critical.

Nutrients

Nutrients are necessary for productive aquatic ecosystems, but in high concentrations, they can adversely affect aquatic life through excessive plant growth in streams, lakes, and coastal waters, leading to depleted dissolved oxygen, and fish kills. Nutrient concentrations in water generally are related to land use in the upstream watershed or the area overlying a ground-water aquifer (Meuller and Helsel, 1996).

Total nitrogen loadings from rivers to coastal estuaries increased from 1900-1994 as a result of increasing use of nitrogen-based fertilizers, the increase in wastewater from municipal and industrial sewage, increased use of de-icing salts on roads, and increased atmospheric deposition of nitrogen. Nitrogen is released into the atmosphere from numerous sources, including fossil fuel combustion, agricultural fertilizers, and animal manure. Large amounts of municipal and industrial sewage were released directly to surface waters in the U.S. as late as the mid-1960s (U.S. Department of the Interior 1968). Aquatic concentrations of chloride, and nitrate, increased during the 20th century due to municipal and industrial wastewater discharges (Jaworski and Hetling 1996). Specific conductance and dissolved chloride concentrations increased in rivers in New England over this same period (Strause 1993, Kulp and Bohr 1993, Bell 1993, Toppin 1993, Trench 1996) likely due to the increased use of de-icing salts on roads. The passage of the Federal Water Pollution Control Act in 1972 resulted in significant improvements in wastewater treatment throughout New England. Although wastewater practices are much improved, wastewater discharges and septic system effluent can still affect water temperature and increase nutrient concentrations such as nitrogen in aquatic ecosystems.

Total phosphorus in northeast waters increased until the 1960s for many of the reasons listed above for total nitrogen, but has decreased since then because of a ban on phosphate-containing detergents (Roman et al. 2000). Water quality of three northeast rivers over the last century showed decreasing concentrations of sulfate and total phosphorus; but increasing concentrations of nitrate and chloride (Robinson et al. 2003).

Contaminants

Contaminants, including trace metals such as copper, lead, mercury, zinc, cadmium, and nickel; organic chemicals such as PCBs; polynuclear aromatic hydrocarbons (PAHs); and pesticides all have been found to adversely affect the quality of surface water and sediments in the northeastern U.S. (Maine Department of Environmental Protection, written communication, 1992). Contaminants accumulate in sediments, are consumed by bottom-feeding organisms, and then work their way up the food chain. Contaminants inhibit the growth, reproduction, and immune systems of aquatic organisms.

Anthropogenic sources of contaminants include industrial effluent, municipal wastewater, runoff from agricultural, urban and forested areas, and atmospheric deposition. Human activity speeds the rate at which naturally occurring metals leach into the environment. Concentrations of lead, mercury and zinc within stream-bottom sediments were positively correlated with urban land use in the Hudson Connecticut, Housatonic and Thames River Basins from 1992-1994 (Breault and Harris 1997, Wall et al. 1998). PCBs were used in electrical equipment until the 1970s and now persist in stream bottom sediments and biota. PAHs , released into the atmosphere by the incomplete combustion of wood, coal, petroleum products, were also found in stream bottom sediments in the Hudson River Basin, and were found to be correlated with the location of current or historic point sources(Wall et al. 1998).

Pesticides can enter surface water bodies through overland runoff or enter groundwater through infiltration. Concentrations and types of pesticides detected in New England streams depended upon landuse (Garabedian et al. 1998). Diazinon was most often detected at the urban sites while atrazine, metolachlor, and simazine were most frequently detected at sites draining agricultural land. Atrazine was detected at 88 percent of the agricultural sites, was frequently detected in combination with other pesticides, and was the most commonly detected pesticide overall. The high percentage of insecticides detected in urban basins reflects the use of these products on lawns. While wide spectrum pesticides such as DDT have been banned in the U.S., contemporary insecticides are soluble in water and can be toxic to fish. Herbicides, while less toxic to fish, can kill aquatic plants (Welsch 1992). Pesticides degrade slowly, accumulate over time, and can be detected in fish tissue even when the concentrations are too low to be detected in stream bottom sediments.

Hydrologic Alterations

Hydrologic alterations have many causes, including increases in impervious surface area associated with development; installation of culverts; water withdrawals and discharges; the installation of, and water storage and release from impoundments; and straightening and/or confining a channel within an urban area. These alterations can directly affect the aquatic flow regime, and water quality. Alterations can also affect geomorphology over the long term by dampening peak flows, changing patterns of aggradation and degradation, constricting a meandering channel, and causing local scour. Hydrologic alterations such as impoundments can restrict the movement of aquatic organisms.

Although soil erosion is a natural aquatic process, human activities can accelerate erosion to the point where it is harmful to ecosystems. Excessive suspended sediments can block sunlight and impair photosynthesis; reduce visibility and the ability of fish and other organisms to feed; raise water temperatures and reduce dissolved oxygen; clog and damage filter feeders and fish gills. Human activities which accelerate erosion include the creation of impervious areas which increase the volume and speed of stormwater runoff and erode stream banks. Construction and forestry projects that leave the soil exposed can also accelerate erosion.

Beavers can also cause hydrologic alterations in aquatic ecosystems. They create wetlands and marshy areas that provide habitat for hundreds of species by building dams and engineering wetlands. The near-elimination of beavers by the beginning of the 20th century led to a drying of wetlands and an expansion of meadows and forests to the detriment of marshy species. Beaver ponds and dams function as water filters that capture silt and pollutants, and result in improved water quality downstream. Despite beavers' reputation for causing flooding, their marshes help buffer adjacent landscapes against the effects of flash floods. Their network of channels, dams, and sloughs slows and holds water in the landscape longer, insulates areas from drought, and recharges underground aquifers. (Wilkinson, T., National Parks Conservation Association, written communication, 2004). Despite the many positive affects of beaver engineering, beavers create challenges for park managers when they occur at an excessive level. Beavers topple trees, flood roads, crops, and woodlands, create impoundments, flood riparian areas, and alter riparian vegetation.

Invasive Species

The presence and persistence of invasive and/or exotic species is a serious issue at all NETN parks. Invasive species can displace native species in wetlands and riparian areas. Invasive plants, also contribute to the channeling (narrowing and deepening) of streams, and the eutrophication and depletion of dissolved oxygen of lakes and ponds, Invasive exotic species can also profoundly affect the visitor experience, by changing the quality of the water used for swimming, boating, fishing, and drinking. The most prolific invasive species in freshwater aquatic habitats are common reed (*Phragmites australis*), purple loosestrife (*Lythrum salicaria*), and curly pondweed (*Potamogeton crispus*).

Visitor Use

Visitor use could be one of the most important stressors acting within park boundaries of NETN parks. The NPS aims to preserve unimpaired the natural and cultural resources and values of the national park system for the enjoyment, education, and inspiration of this and future generations as a part of its mission. The mission of preserving unimpaired while also having the public enjoy, be educated and inspired is complex. Stressors to freshwater aquatic resources related to visitor use include the extraction of natural resources (such as fish), erosion stemming from multiple uses, road runoff and contamination stemming from the many roads that allow visitor access within the parks, and the introduction of invasive species carried in by visitors. While little research has been done relating visitor use to the condition of the aquatic resources, the potential for this use to cause stress to natural ecosystems is great, especially when compounded with other stressors such as climate change and invasive exotic species.

Intertidal resource conceptual narrative

In this section we present the intertidal resource conceptual ecological model as a diagram (Figure 4) accompanied by the following narrative describing our current understanding of intertidal system components and their interactions.

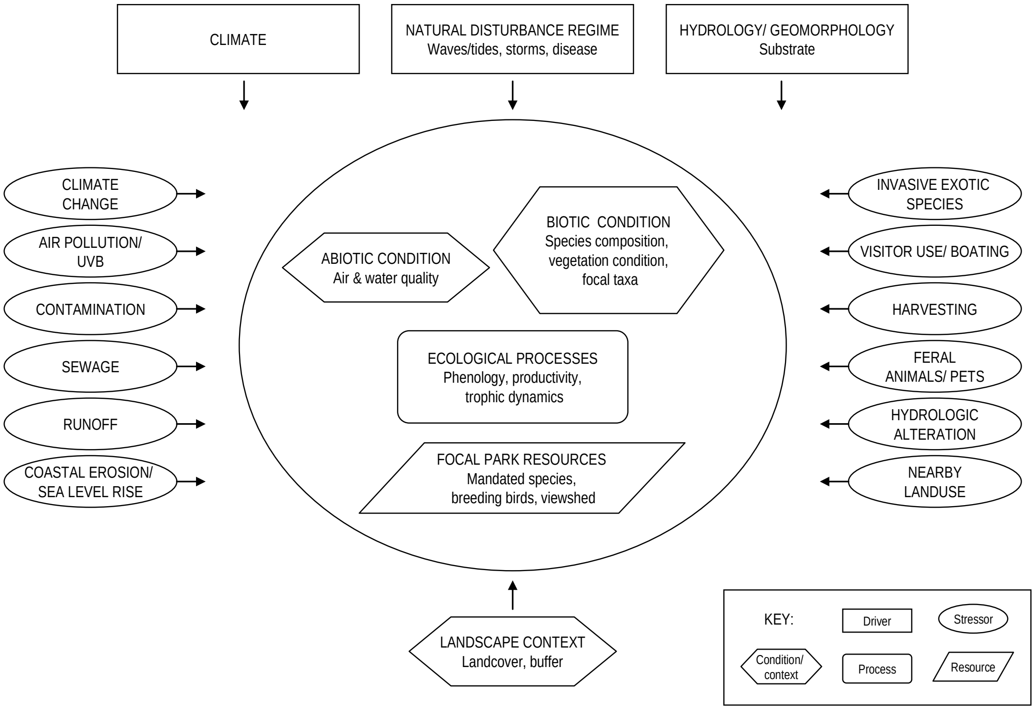

Figure 4. Intertidal conceptual model diagram.

Ecological systems

Intertidal systems are present in two of the 11 NETN parks - Acadia National Park and the Boston Harbor Islands NPA. Unlike intertidal systems further south, the systems in these northeastern parks are primarily rocky intertidal systems, with limited areas of mudflat, sand and salt marsh systems (Table 1). This is due to the geologic history of New England (Bertness 1999). Pleistocene glaciation scoured sediments from New England shores, so the New England coast lacks the extensive barrier beach and salt marsh habitats which develop from sediment accumulation and are common south of Boston Harbor.

Rocky Intertidal

The rocky intertidal systems which dominate the New England coast are characterized by strongly fluctuating physical conditions, caused by tides, and creating stark patterns of vertical zonation from the low to high tide zones. The rocky substrate offers less respite from extreme temperatures, desiccation, and buffeting waves than soft-sediment shores, and thus favors algae and invertebrates which can withstand these physical challenges. The rocky intertidal food chain is supported by high plankton productivity, harvested by filter-feeding barnacles and mussels, and also by benthic algae, consumed by herbivorous snails and urchins. Dominant predators include shell-drilling snails and starfishes in open-coast habitats, and crabs in bays and estuaries. The intertidal zone also provides food for many species of birds, and haul-out habitat for harbor seals. Native species composition of northeastern rocky intertidal habitats is relatively depauperate due to Pleistocene extinctions caused by climatic extremes and changes in sea level associated with glaciation (Stanley 1986). This has left these systems vulnerable to exploitation by invasive exotic species. Many classic experiments in community ecology have been undertaken in rocky intertidal habitats because their zonation and relatively simple communities make them attractive for study (Connell 1961, Lubchenco 1978, Roughgarden et al. 1985); thus patterns of disturbance, recruitment, competition, and trophic relationships are relatively well understood in these well-studied systems. Within the upper intertidal zone, physical stress is a primary driver of ecological pattern, while life in the lower intertidal zone is controlled by consumer pressure and competition.

Mudflats

Intertidal mudflats form in protected areas along the coast where diminished water movement allows the accumulation of fine sediments. In contrast to rocky intertidal habitats, organisms inhabiting mudflat systems interact more dynamically with the substrate, burrowing or growing into the mud and respectively increasing or decreasing habitat stability by doing so. Sediments in mudflats possess strong vertical biogeochemical gradients due to subsurface anoxic conditions caused by submersion. Often, a sharp boundary demarcates the anoxic zone, below which anaerobic decomposition processes and chemotrophic bacteria prevail (Howarth and Teal 1980). Intertidal mudflats often support large predator populations - birds, fishes and crabs which feed on worms, clams, and small crustaceans. Food supply in mudflats is strongly linked to water movement processes, which supplies both plankton for filter-feeding bivalves, and detritus for deposit-feeding organisms.

Salt Marshes

Like mudflats, salt marshes also develop in protected coastal habitats, often the mouths of estuaries, where fine sediment accumulation enables colonization by halophytic vegetation. Salt marsh systems are successional, beginning with colonization by smooth cordgrass, *Spartina alterniflora*, which binds additional sediment to create higher marsh habitat above tidal influences and subject to colonization by additional species (Redfield 1972). Disturbance from winter ice-scour is common in northern salt marshes, and resets this successional development. Like rocky intertidal systems, salt marshes exhibit strong elevational zonation due to gradients of physical stress and competition, though in salt marshes physical stressors (from anoxia and salt) drive ecological patterns at lower elevations while competition dominates at higher elevations more suitable for plant growth. Plant survivorship in salt marshes has been shown to be enhanced by positive neighbor interactions, such as increased oxygenation of soils and reduced salt stress, and these effects are considered important in these systems (Howes et al. 1986, Hacker and Bertness 1999). Salt marsh food chains are typically detritus-based, with consumers primarily feeding on plant detritus. Salt marshes provide numerous benefits, serving as protected nursery grounds for many species of fish, shrimp and crabs, providing feeding and nesting area for birds and mammals, buffering shorelines from flood and storm damage, limiting erosion, and reducing coastal nutrient loading by providing sinks for excess nitrogen and sulfur. Despite these numerous benefits, New England salt marshes have been extensively grazed, drained, filled, developed and otherwise altered (Dreyer and Niering 1995). Of particular note are coastal road and rail corridors built during the early 20th century, which filled many marshes and isolated many remaining marshes from coastal water flows.

Ecosystem drivers

Climate

Basic climate data is critical for understanding and interpreting intertidal zone species change. Important measures include air and water temperature, precipitation, and wind speed and direction. In addition, snow/ice depth within the intertidal zone may significantly control species composition and abundance. Storms and tides are important agents of disturbance within the intertidal zone. The frequency and duration of storm events should be monitored via measures of climate or landscape pattern. Wave energy data may be available from existing offshore wave gauges, deployed by agencies such as NOAA.

Substrate composition is the primary determinant of community type within the intertidal zone, and thus is also an important indicator of biotic change. While bedrock and boulder substrates exhibit little change over time, cobble, gravel, sand and mud substrates change both seasonally and over the long-term, in response to storms and sometimes human use.

Ecosystem condition

Water Chemistry

Water chemistry is directly related to changes in floral and faunal distribution within the intertidal zone. This basic information supports the establishment of relationships between physical and biological processes, and informs management decisions. Important measures of water quality in the intertidal zone include water temperature, conductivity/salinity, and water clarity.

Intertidal Flora and Fauna

Determining and monitoring species richness, abundance and distribution of intertidal macro-algal vegetation is critical to understanding status and trends of the intertidal zone. Monitoring should focus on attached flora, which form the base of the community within the rocky intertidal zone. Much of this vegetation is perennial; some, like *Ascophyllum* can live for decades and exhibit low recruitment and slow growth (Bertness 1999). Ephemeral green algae such as *Ulva* flourish in high nitrogen waters and thus indicate eutrophication. Invasive species like *Codium* are invading the northeast, and may be indicative of climate change and other disturbance.

Species richness, abundance, and distribution of intertidal fauna can also be valuable information for long-term monitoring, though fauna exhibit higher spatial and temporal variability than flora. Monitoring of select intertidal fauna may be a particularly useful indicator of visitor trampling. Potential faunal groups that would be useful indicator taxa include key predators, such as gulls, Eider ducks, and green crabs; mussel populations, which may be indicative of trophic change or offshore disturbance from salmon pens; or the invasive Asian shore crab, which is an aggressive invader capable of thriving in cold Maine temperatures. Periwinkles may also be useful taxa for monitoring. Intertidal faunal monitoring should consider seasonal patterns.

Landscape context

Community Type

A well-documented map of intertidal community types, including substrate type and biotic assemblage, is essential for understanding current conditions and monitoring long-term change.

Stressors

Pollution, invasive exotic species and harvesting are the most serious threats currently facing New England intertidal systems, though sea level rise and shoreline erosion are expected to seriously threaten these systems over the next century. Visitor impacts are also important stressors on NETN intertidal systems.

Pollution

Pollution from many sources significantly impacts intertidal systems. Oil pollution, from urban and suburban runoff and from tanker spills, is a chronic problem (Suchanek 1993). Some seaweeds and many crabs, gastropods and amphipods are very sensitive to oil pollution. Sewage runoff is likewise a pervasive nearshore stressor, which can cause coastal eutrophication and toxic algal blooms that negatively affect native species (Valiela et al. 1992). Toxic, anti-fouling paints routinely applied to the undersides of boats are another widespread, chronic stressor; these paints leach into nearshore waters and affect many intertidal organisms (Gibbs et al. 1988).

Invasive Species

Invasive exotic species are widespread within New England intertidal systems. The native species composition of these systems was depleted by extinctions caused by Pleistocene glaciation (Stanley 1986), leaving these systems particularly vulnerable to invasion by exotic species. Historic and modern shipping practices have supplied a steady influx of invaders, including some of the most common species now encountered (Carlton 1985). These factors have drastically altered New England intertidal community composition over the last few hundred years and probably caused many local extinctions, but we lack knowledge of intertidal community composition prior to European exploration and settlement. Within New England salt marshes, the exotic reed *Phragmites australis* has been particularly destructive, outcompeting native marsh plants and altering habitat. New invasive exotic species continue to arrive and spread.

Harvesting

Throughout the history of human settlement in New England, humans have harvested a wide variety of intertidal organisms. While some species are now protected from over-harvesting, collection of many species continues. Shellfish and bait worms are harvested from soft-bottom flats within both ACAD and BOHA. Rockweed and knotted wrack (*Fucus* and *Ascophyllum*) are harvested for lobster-packing. In addition, many species are commercially harvested from the subtidal zone, immediately adjacent to the intertidal zone; these species include sea cucumbers, lobsters, and sea urchins. Some data describing the intensity of harvesting activity could be compiled from exiting data collected by local regulatory agencies, such as state agencies and town shellfish wardens.

Water Level

Sea level controls the distribution and spatial pattern of intertidal habitats, thus as sea level rises, the boundary of intertidal habitat types will shift. Currently, sea level is rising about 2-4 mm/yr along the New England coastline due to global warming, and this rate of change is predicted to accelerate. Sea level data can be compiled from data collected by existing tide gauges in Boston and Bar Harbor operated by NOAA. In addition to sea level rise, shoreline erosion can cause change in the distribution of intertidal communities by loss of physical habitat via movement of intertidal sediment. Shoreline erosion is caused by a variety of natural and anthropogenic forces, including storm wave energy and boat wakes. Shoreline change could be monitored in part as change in mapped distribution of intertidal community types.

Visitor Use

Finally, the rocky and sandy intertidal zone is a frequently visited habitat and often the focus of park-led interpretive tours at both ACAD and BOHA. Visitor use at both these parks can cause substantial trampling and removal of resources. In order to truly understand biotic change within the intertidal zone, it will be important to monitor visitor use, and more specifically, visitor intensity, location, and activity, such as walking, boating, or recreational shell-fishing. Trampling and other visitor impacts are likely to be localized within areas accessible to parking or ferry. Some data on visitor use may be available for compilation from existing park efforts.

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A.

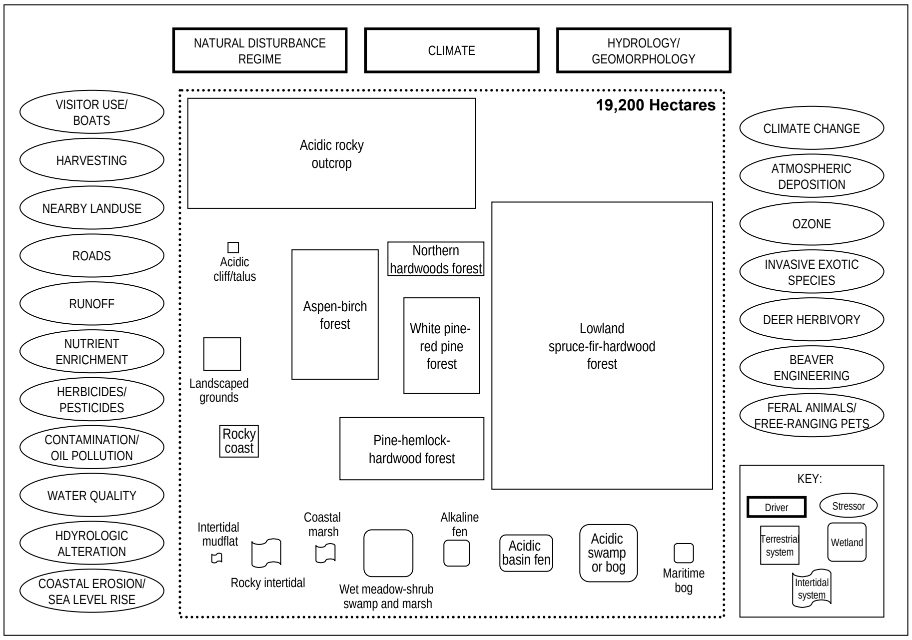

Figure 5. Acadia NP Terrestrial Conceptual Model diagram.

B.

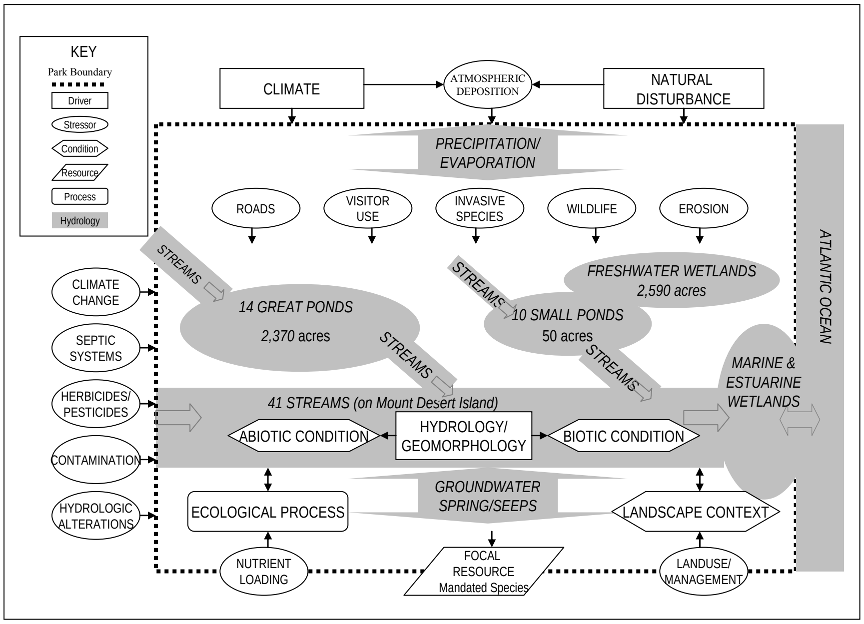

Figure 6. Acadia NP Aquatic Systems Conceptual Model Diagram.

A.

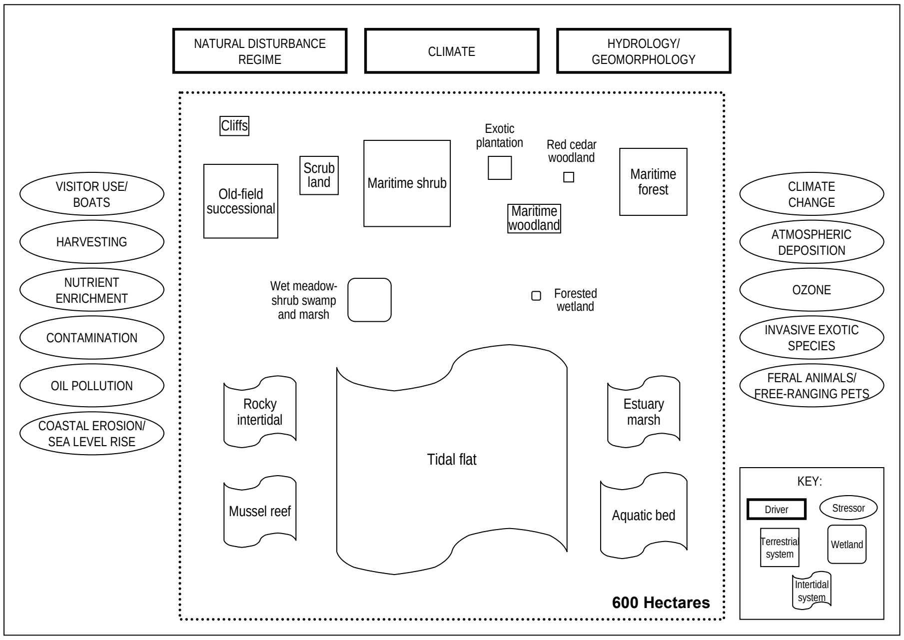

Figure 7. Boston Harbor Islands NPA Terrestrial

B.

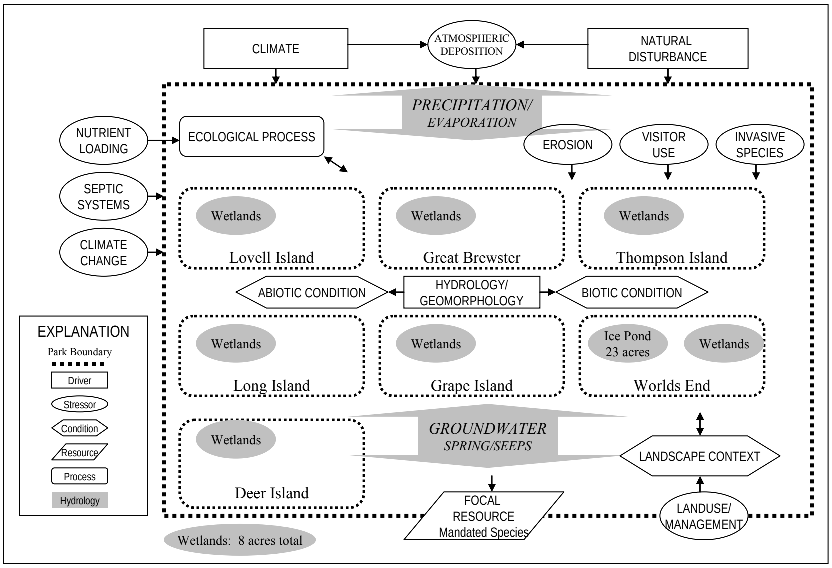

Figure 8. Boston Harbor Islands NPA Aquatic Systems

A.

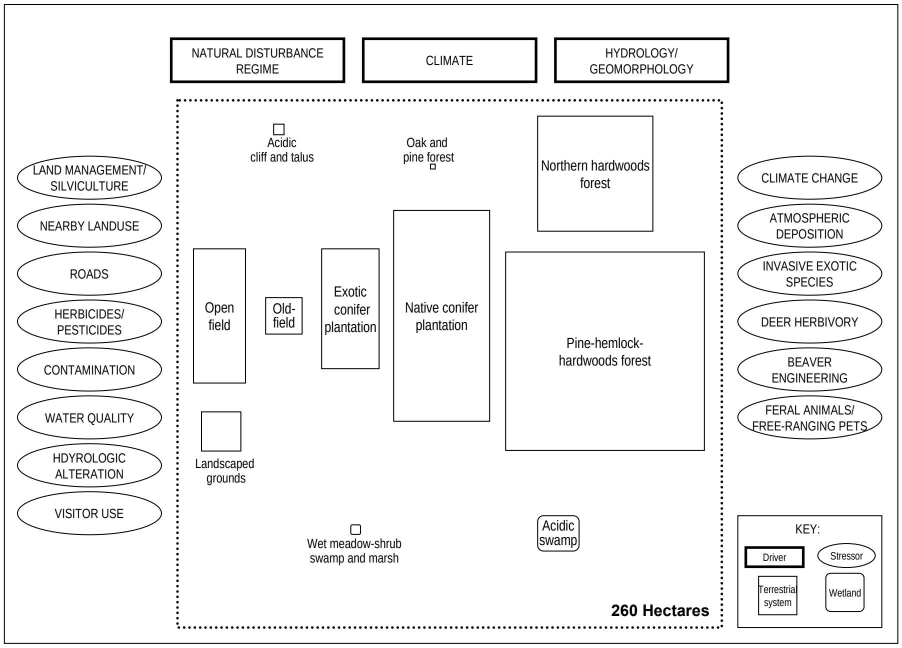

Figure 9. Marsh-Billings-Rockefeller NHP Terrestrial

B.

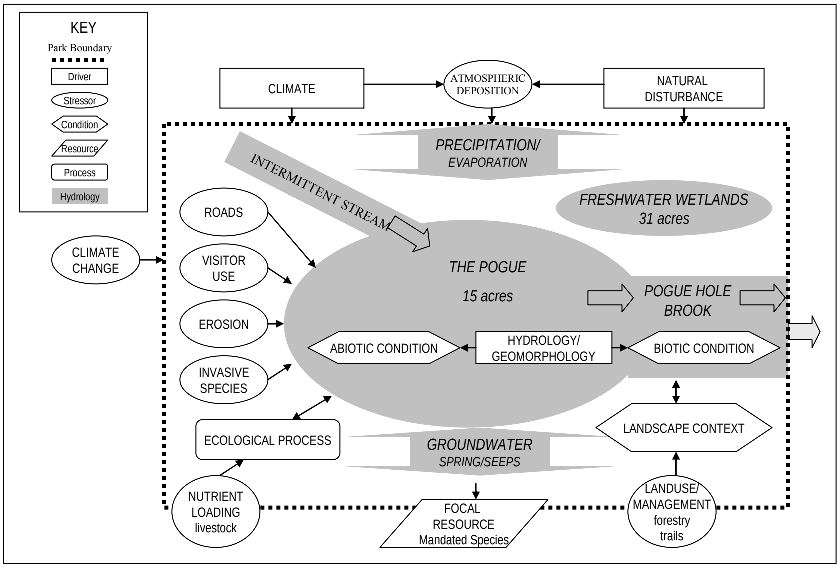

Figure 10. Marsh-Billings-Rockefeller NHP Aquatic Systems

A.

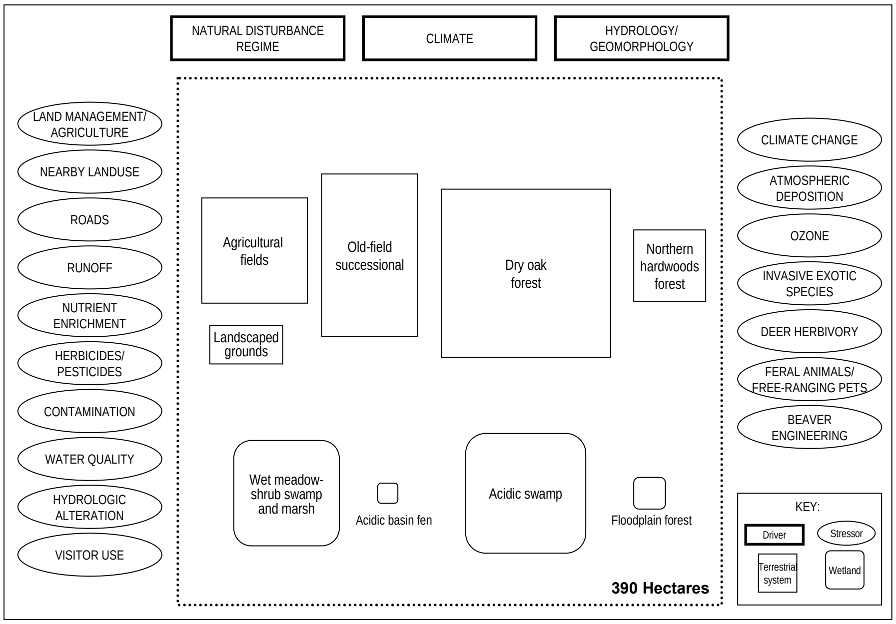

Figure 11. Minute Man NHP Terrestrial

B.

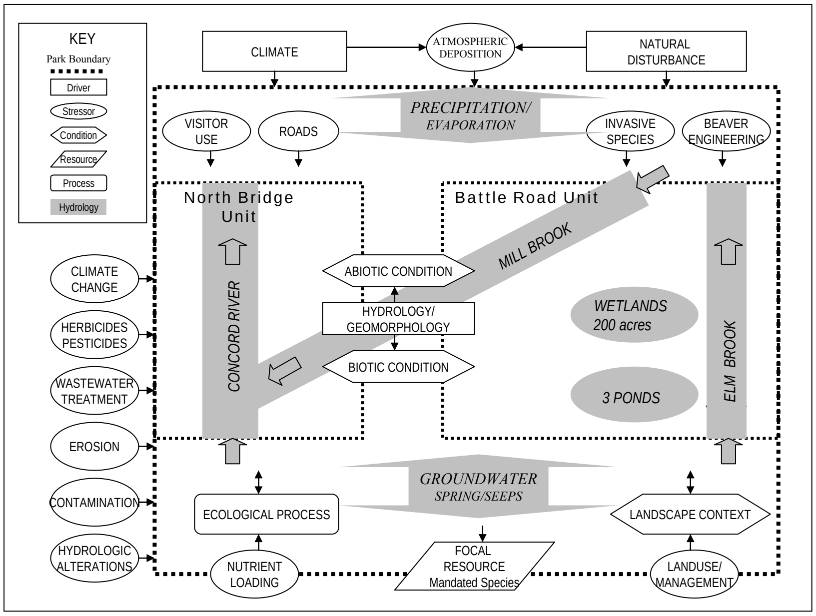

Figure 12. Minute Man NHP Aquatic

A.

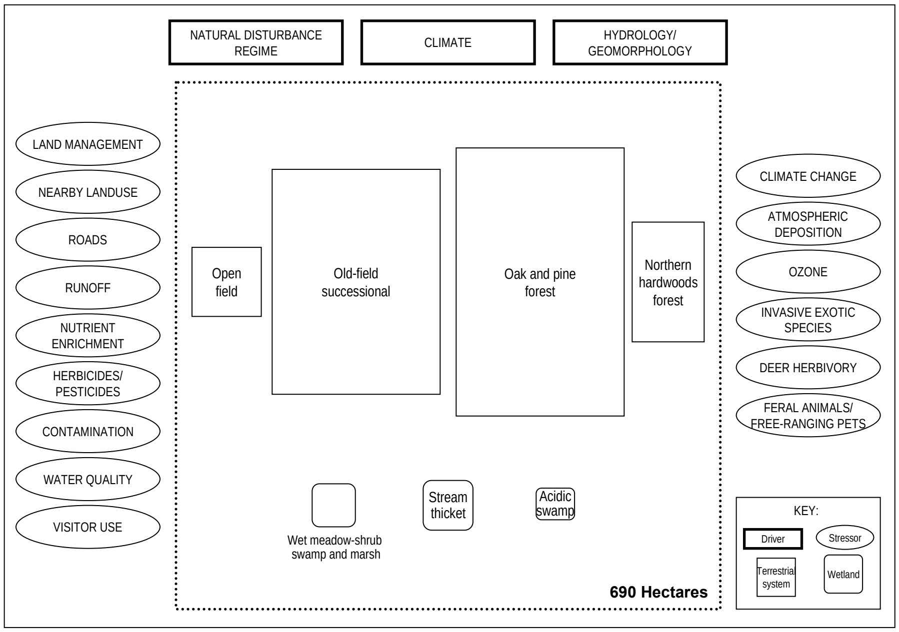

Figure 13. Morristown NHP Terrestrial

B.

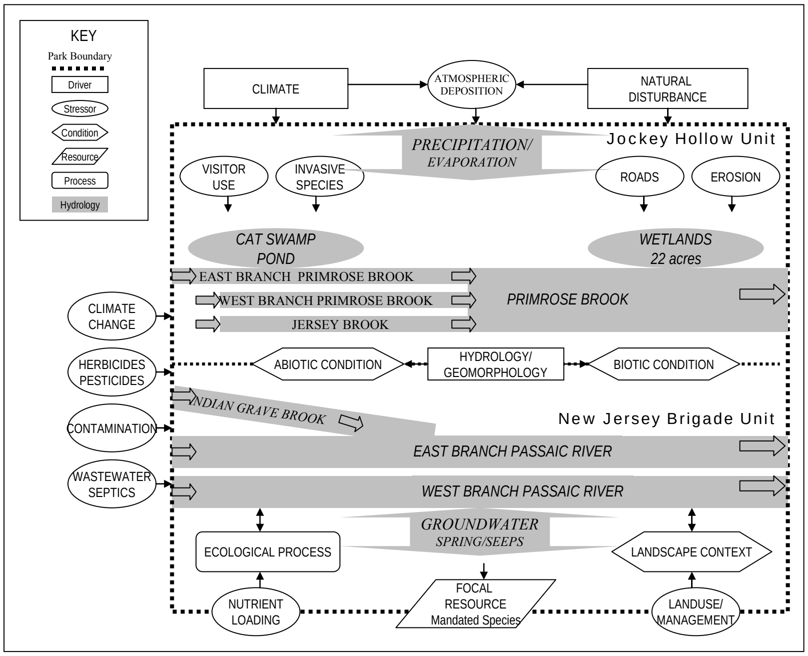

Figure 14. Morristown NHP Aquatic

A.

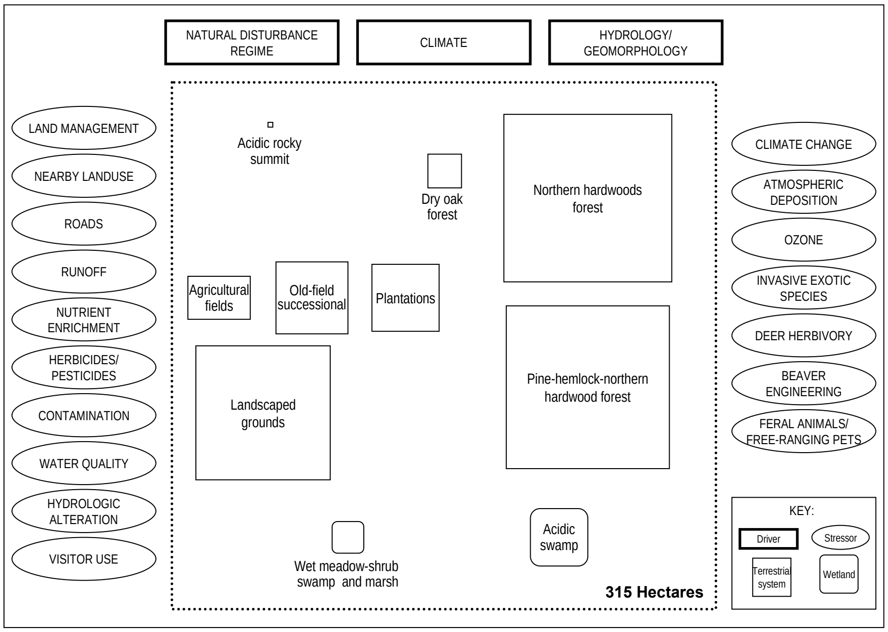

Figure 15. Roosevelt-Vanderbilt NHS Terrestrial

B.

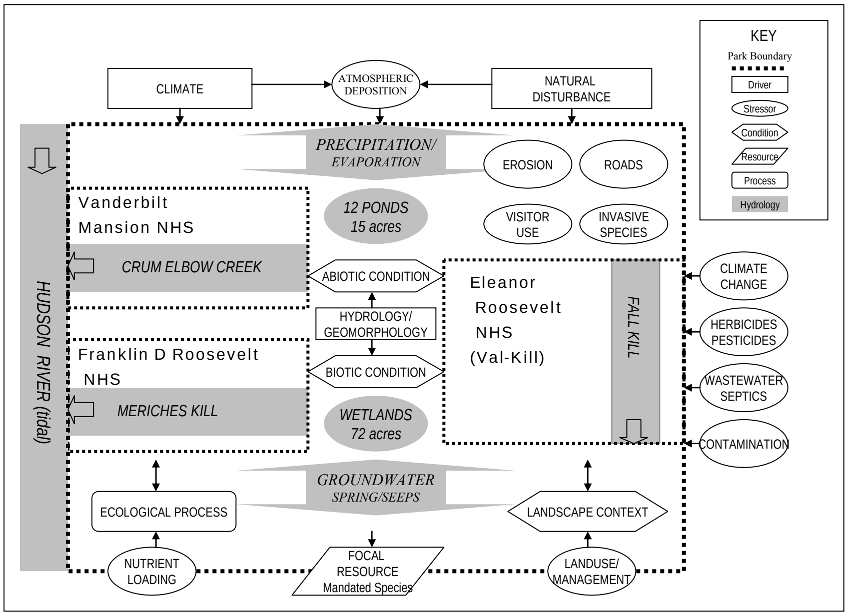

Figure 16. Roosevelt-Vanderbilt NHS Aquatic

A.

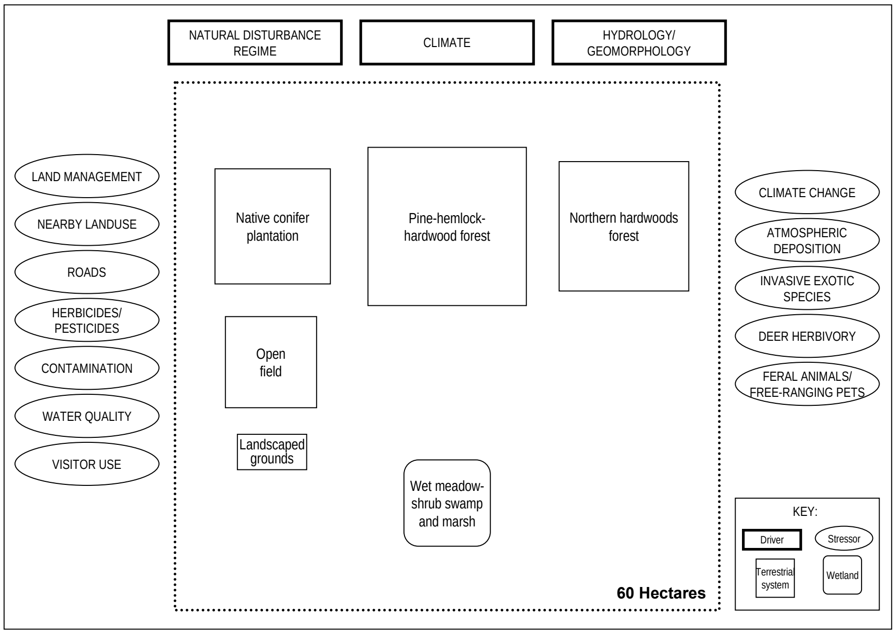

Figure 17. Saint-Gaudens NHS Terrestrial

B.

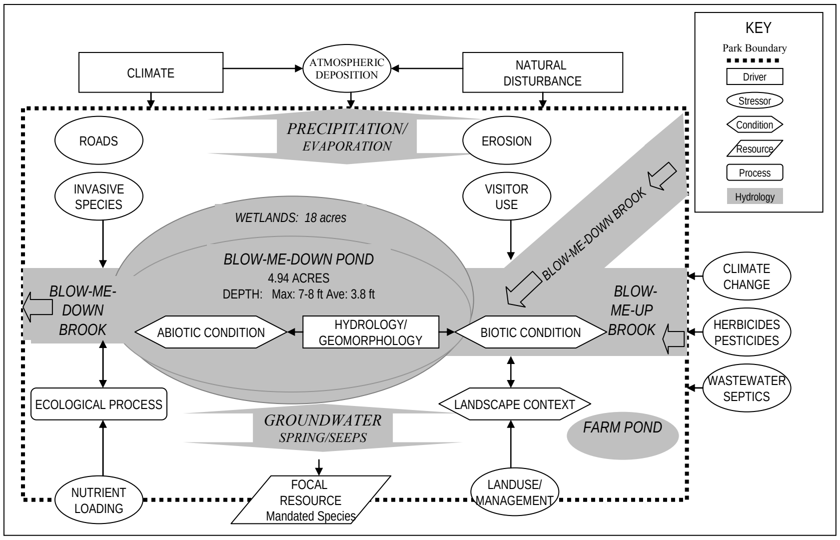

Figure 18. Saint-Gaudens NHS Aquatic

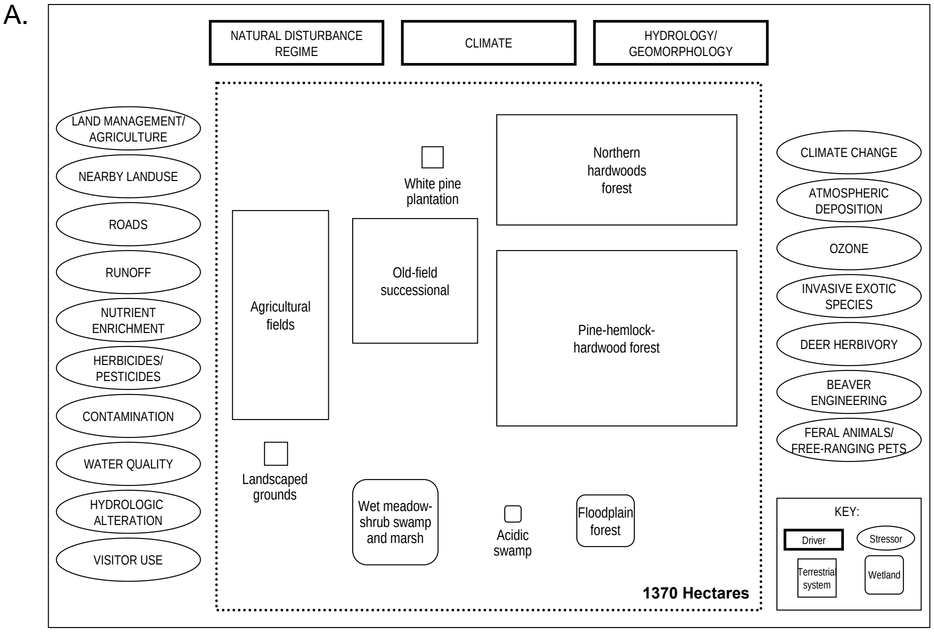

Figure 19. Saratoga NHP Terrestrial

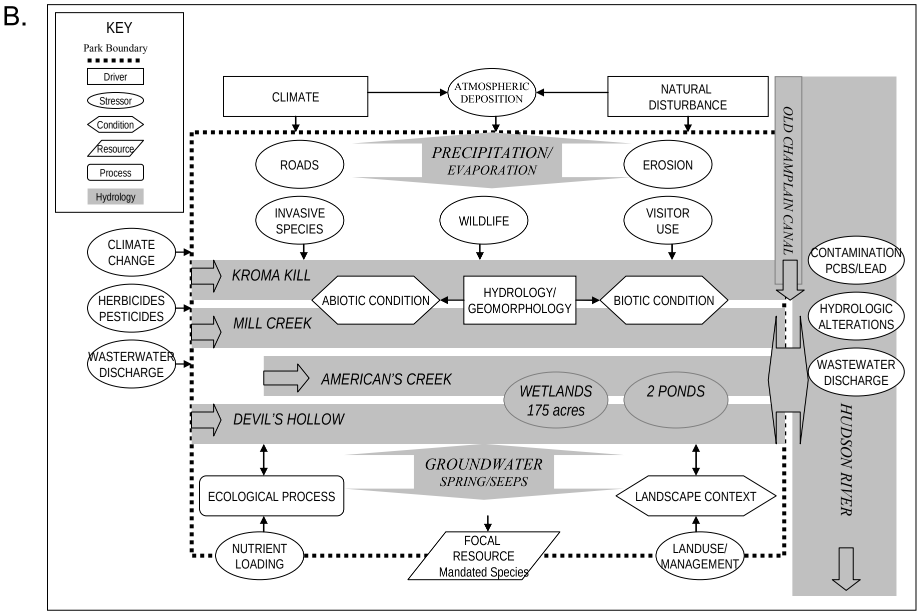

Figure 20. Saratoga NHP Aquatic

A.

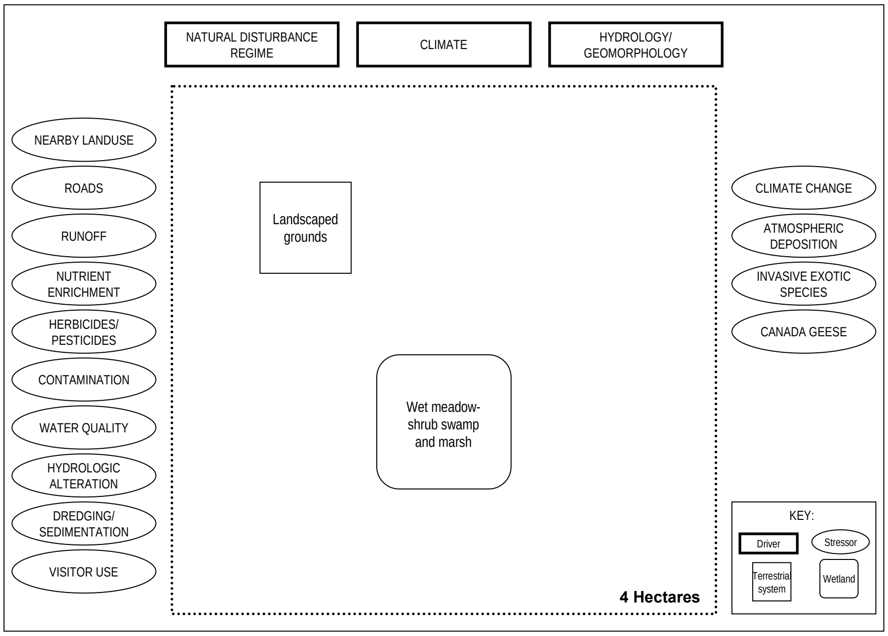

Figure 21. Saugus Iron Works NHS Terrestrial

B.

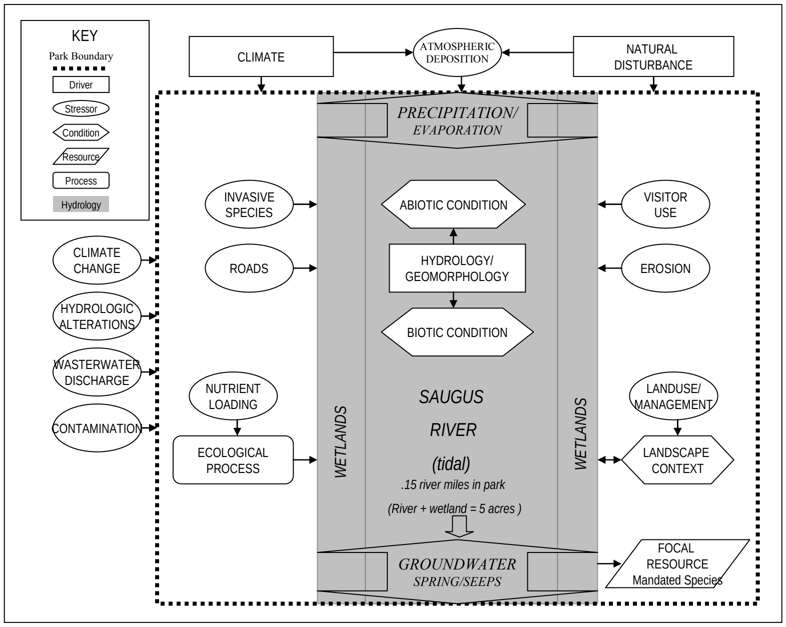

Figure 22. Saugus Iron Works NHS Aquatic

A.

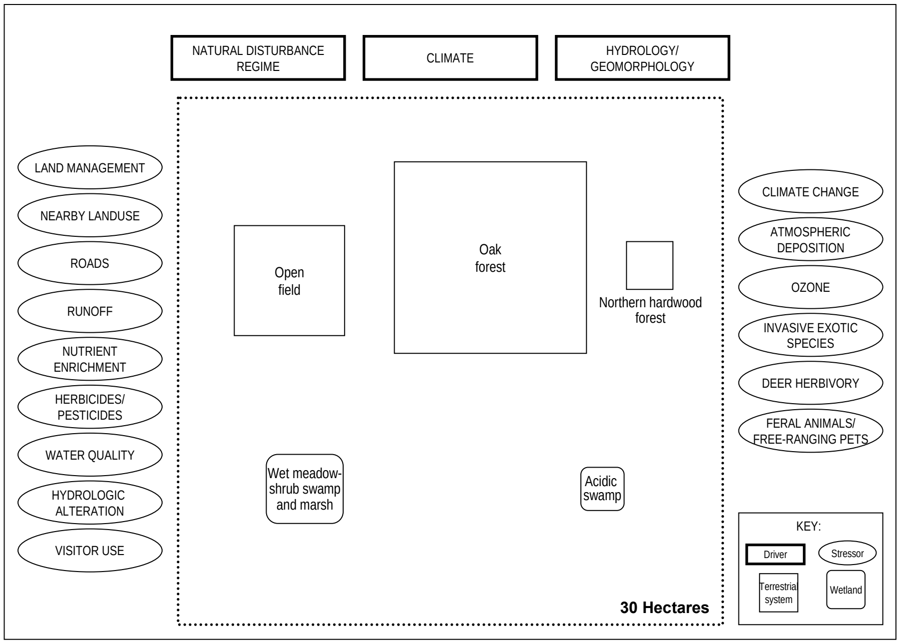

Figure 23. Weir Farm NHS Terrestrial

B.

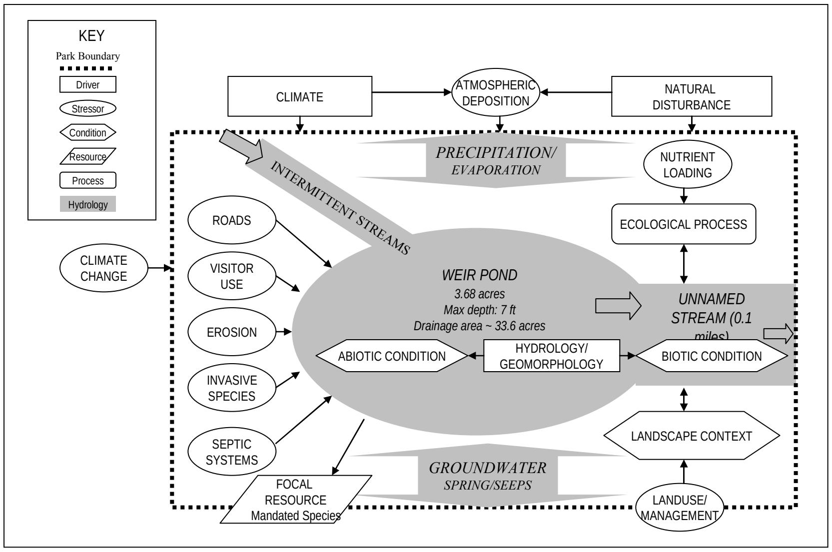

Figure 24. Weir Farm NHS Aquatic