FIRE IN THE PINES: A LANDSCAPE PERSPECTIVE OF HUMAN-INDUCED ECOLOGICAL CHANGE IN THE PINELANDS OF NEW JERSEY

by

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ABSTRACT OF THE DISSERTATION

FIRE IN THE PINES: A LANDSCAPE PERSPECTIVE OF HUMAN-INDUCED ECOLOGICAL CHANGE IN THE PINELANDS OF NEW JERSEY

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Effects of urban land-uses have long term implications for the structure and function of natural ecosystems that may extend far beyond the land-use itself. Specifically, natural disturbance and succession in forest ecosystems have been highly altered by human-caused land-use and fire frequency changes. Changes to forest community structure and composition can affect the long-term sustainability of areas such as the New Jersey Pinelands, a fire-dependent ecosystem. By combining historic maps of fire frequency and land-use change, I assessed the effects of human development patterns on fire and forest composition in the Pinelands. These assessments showed lower fire frequency and higher transitions from pine to oak forest cover in close geographic proximity to altered land. Additionally, I investigated our ability to detect the effects of fire on water quality measures using data from gauged watersheds. No significant effects of fire could be determined due to a lack of water quality data associated with wildfires in space and time. I used a spatially-explicit forest disturbance and succession model to investigate how increasing levels of altered land and changing fire regimes may affect forest composition

in the future. Additionally, I added climate change to disturbance and succession modeling to incorporate this additional forcing on fire and forest composition. These scenarios showed an overwhelming trend toward oak dominated forest within 100 years, except in the unique pine plains area, where pine species still dominated. The potential of this type of dramatic shift from pine to oak cover represents a radical departure from current forest composition and needs to be addressed by managers of the Pinelands National Reserve in order to maintain the essential Pinelands landscape. Modeling the potential influences of current and future altered land as well as changes in fire regimes in our study area elucidates the degree to which fire and climate disturbances may alter forest composition.

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DEDICATION

To my two Dads, one lost long ago, the other so recent. Robert Lawrence Parker gave me my love of all things spatial and the inspiration to follow my dreams to wonderful adventures. Karl-Heinz Kopf made the world and its opportunities come alive with his amazing presence, love, support, and never-ending confidence in me. His intelligence inspired me to want to 'know it all'. He was always saying he took me camping "one too many times". I'm pretty sure it was just about right. Thanks Dad.

To the sweetest man I'll ever know, David La Puma…thank you for helping me accomplish what felt impossible...and for being there for me always and for being there for our two beautiful girls Corinna Wren and Rosa Lark…especially when I needed all the help I could get.

To my little La Puma girls…your smiles and triumphs got me through this.

To my Mom, the most patient and best friend I've ever had…I know you had more to do with this than anyone.

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INTRODUCTION

Traditional ecological studies have often viewed natural systems in isolation; however, increasingly, researchers are necessarily moving toward a coupled human-environment framework (Millenium Assessment 2005). This new paradigm incorporates feedbacks such as how ecosystems affect humans via ecosystem services and how humans affect ecosystems through various human disturbance actions. Three major spatial and temporal scales need to be considered when defining the changes that might occur to an ecosystem under the coupled human-environment framework: the human influence scale, the abiotic process scale, and the biotic process scale. These scales are temporally and spatially interdependent; therefore it is essential that all perspectives are considered when attempting to assess ecosystem change. Feedbacks between how the environment affects us, through providing resources or ecosystem services, and how we affect the environment, through disturbance and management decisions are important aspects of the coupled human-environment system. Using this type of landscape ecology perspective, this dissertation focuses on human caused disturbance and the consequences of such disturbances on a specific ecosystem. Specifically, we look at how human altered landscapes indirectly affect disturbance regimes and ultimately, the range and extent of the ecological consequences of those changes. Human-caused variation in forest succession due to disturbance has been shown to affect forest composition (Dale et al. 2001) and sustainability (Kimmins et al. 2007) and includes changes to ecosystem services such as fire safety (Bar Massada et al. 2009), water quality (Carignan et al. 2000), and carbon sequestration (Hurteau et al. 2008). Since multiple feedback loops exist between the human, abiotic and biotic scales, the range of viable ecological services is likely to change under future land-use and fire regime disturbance scenarios.

The Pinelands of New Jersey represent a unique ecosystem and landscape that reflects a long history of anthropogenic influence, such as logging and fire ignitions, and ecosystem services, such as clean water supplies, endemic species habitat, and carbon sequestration. I used the New Jersey Pinelands as a case study to investigate gross changes in human altered land and fire regimes as well as differences in forest composition and water quality resulting from these disturbances. I focused on the Mullica River and Barnegat Bay watersheds as the two watersheds that drain to the Jacques Cousteau National Estuarine Research Reserve (JCNERR), a National Oceanic and Atmospheric Administration (NOAA) research site for which I obtained several years of funding to conduct this research. Additionally, these two watersheds comprise a large part of the Pinelands ecosystem as delineated by the Pinelands National Reserve created in 1978 (Chapter 1, Figure 1) and designated as an International Biosphere Reserve by the United Nations Educational, Scientific and Cultural Organization (UNESCO) in 1988.

Changes to forest succession near human altered land due to fire suppression has been assumed in the Pinelands, but has not yet been investigated in a scientific manner at the landscape scale. This area has been termed the wildland-urban interface (WUI), where varying levels of population densities are interspersed or interface with natural wildlands (Radeloff et al. 2005). The goal for this research was to provide a clear and scientific understanding of the relationship between landscape level fire and forest composition and water quality in the Pinelands. The challenge in this ecosystem lies in understanding community and ecosystem level interactions while meeting long term management challenges of maintaining fire disturbance and protecting human life and infrastructure. Understanding how the changing patterns of fire and human development and their interactions might shape the Pinelands is critical in creating science-based management plans.

To accomplish these goals we need a better understanding of:

1. the spatial-temporal patterns of fire disturbance and the potential influence of the wildland/urban interface (WUI) in altering the patterns of both ignition and fire size (*the GIS analysis*);
2. relationships between historic fire and water quality data (*statistical analysis*); and
3. the effects of increases in altered land and associated changes to fire regimes as well as climate change forcings on forest composition (*the modeling scenarios*).

My research was designed to help address the following fundamental questions. As human development continues and the Pinelands becomes more fragmented, will the WUI act as a buffer to preserve the Pinelands as a functioning ecosystem; or will the effects of development fundamentally change the entire Pinelands ecosystem through altered disturbance regimes across the landscape? Will the essential ecosystem service of excellent water quality be affected by ash deposition leading to chemical changes in pH and specific conductance; or will the effects of fire be a minor component of water quality changes? As climate influences increase, will higher temperatures and CO_2 concentrations markedly alter potential forest composition trajectories due to changes in fire regimes or aboveground net primary productivity (ANPP)? This project aims to understand the extent and severity of change resulting from interacting disturbances in the Pinelands ecosystem. Because of its spatial arrangement - islands of natural areas surrounded and interspersed with development creating a complex WUI (Radeloff et al. 2005) - the NJ Pinelands serves as a model for other biosphere reserves adjacent to rapid urbanization.

My dissertation is comprised of 4 chapters. First I attempt to characterize present trends in altered land, fire, forest composition and water quality; then, understand which trends could be successfully modeled across the landscape. In Chapter 1, I determined landscape level historic patterns of fire in the Pinelands in relation to proximity to human-altered land in order to assess how the changing fire regime has affected forest composition change across the landscape. In Chapter 2, I investigated our ability to assess the proximity and size of fires in relation to associated water quality data and relationships between fire and water quality parameters such as pH, SC, and turbidity for two of the major watersheds in the Pinelands of NJ. In Chapter 3, I used a landscape disturbance and succession model to understand the range of possible effects of changing extents and amounts of altered land and modern fire regimes on forest composition, concentrating on pine versus oak cover. In Chapter 4, I added climate change to my altered land and fire scenarios to assess the effects of increasing temperatures and carbon dioxide as added disturbance forcings in our ecosystem. Chapters are written in the form of stand alone manuscripts in anticipation of publication in peer-reviewed journals. The writing style is in first-person plural in anticipation of submission to academic journals and indicating the significant contributions of my advisor Rick Lathrop to all four manuscripts in helping to shape the focus of the analyses and conclusions therein. Additionally, Rob Scheller and Steve van Tuyl made significant contributions to initial data from their previous work in my study area and guided my methods through their suggestions for Chapters 3 and 4. My first dissertation chapter is currently under revision for *The International Journal of Wildland Fire*.

Finally, I use the concluding remarks to portray the overall lessons learned from this dissertation research and to highlight the management recommendations that are of special interest to local policy makers.

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CHAPTER 1

Defining the ecological wildland urban interface: relating fire and altered land to forest succession in the Pinelands of New Jersey

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Abstract

Human land development adjacent to protected natural land may result in changes in the magnitude, frequency, and type of natural disturbances critical to preserving the ecological integrity of our protected landscapes. Understanding the complex relationships of coupled human-environment processes is necessary, but first we must assess the spatial extent and intensity of the influence of human land development on adjacent ecosystem processes. We propose a framework to define the ecological wildland urban interface (EWUI) of a natural area. The EWUI area is defined by determining the amount and severity of influence that human altered land has on the ecology of adjacent natural areas. The EWUI is created through human caused changes in disturbance regimes. The Pinelands National Reserve was used as a case study for defining the EWUI as it relates to wildland fire. A major concern in the Pinelands of New Jersey is a shift from pine to oak dominated forest associated with reduced wildfire. Landscape level changes of altered land and forest composition change were evaluated at specific distances from human altered land for two dates for comparison to historic wildfire frequency. Upland coniferous forest transitioning to upland deciduous forest and areas that have been close to altered land for both dates showed similar forest fire frequencies in our study area. Interior areas (those areas farthest from altered land for both dates), displayed fire frequencies similar to upland mixed forest transitioning to upland coniferous forest. Our study shows that fire suppression due to the presence of human altered land created an ecological wildland urban interface area extending up to 480m from altered land. These areas exhibited disproportionate reductions in fire frequency and coniferous to deciduous transitions over a 16 year time period. Thus,

significance of the EWUI is as a metric for characterizing the amount and intensity of the human influence on natural disturbance and ecological processes on protected landscapes.

Introduction

Human caused land cover change has long-term implications for the structure and function of natural ecosystems that may extend far beyond the boundaries of the intended land use change. Specifically, disturbance and succession in forested ecosystems can be highly altered by the presence of adjacent human-caused land use change. Variation in forest succession due to human-induced changes to disturbance regimes has been shown to affect forest composition and persistence as well as ecosystem services such water quality, carbon sequestration, and fire safety with the loss or gain of fire-resistant species (Foley et al. 2005). Forest composition shifts may also affect habitat availability for endemic species or species of concern or cause changes in fire hazard levels near homes. Consequently, evaluating the interactions of land use/land cover (LULC) change and disturbance regimes leading to ecological consequences is a first step in establishing thresholds for ecosystem modeling efforts and conservation management scenarios.

The area where humans reside in or near natural areas is known as the wildland urban interface (WUI), a common term used in fire safety studies. The WUI area is important in the context of fire safety and wildland fire fighting, since if the forests near residential areas are highly flammable, homes are more at risk. The term wildland-urban interface is most commonly defined from a census perspective, using different levels of housing density within a matrix of natural vegetation to determine fire danger in relation to residential density, usually categorized as either interface, or areas containing less than 50% vegetation and within 1.5 miles of contiguous vegetation over 500 ha or intermix, containing areas with greater than 50% vegetation in more sparsely populated areas and near contiguous vegetation over 500 ha (Bar Massada et al. 2009; Radeloff et al. 2005; Zhang et al. 2008). This definition omits non-residential human altered land, including industrial and agricultural areas. From a public fire safety perspective, the residential areas used to define the WUI are sufficient, but they are too narrow to be used for studies on the ecological effects of the human influence on fire regimes and forest ecology. A diversity of human land uses, such as the areas humans inhabit, work in, or are merely present for recreation on a seasonal basis can contribute to disturbance on adjacent natural areas. For example, fire ignitions may occur often near residential housing areas, but are also possible due to activities in commercial/industrial or agricultural areas. We argue there is a need to expand the WUI area to include all human altered land when addressing the ecological effects of human presence. Although Syphard et al (2007) succeeded in addressing the influence of WUI areas on fire frequencies and investigated the influence of fire frequencies on vegetation types, they did not directly relate the WUI to consequences on an ecological process. The relationships between the spatial extent of human influence on fire regimes and the ecological affects those changes have had is an important next step for land managers to move beyond the safety aspects of fire danger in the WUI and address the ecological results of alterations in disturbance in these areas.

A defining research question arises as human land development becomes more extensive or undergoes spatial shifts in configuration. Will the WUI interface and intermix areas act as buffers to preserve core natural areas as desired functioning ecosystems (Wade and Theobald 2009); or will the effects of altered land fundamentally change the ecosystem within the WUI and beyond through long term changes in historic disturbance regimes across the landscape? In lieu of concentrating solely on the area where housing meets residential development for fire safety reasons, we focus on a functional definition of the boundary between human altered land and natural lands incorporating disturbance and ecosystem change (Cadenasso et al. 2003). The focus of this research was to create a framework and provide an example for the concept of an *ecological* wildland urban interface, or EWUI. We define the EWUI as the area potentially impacted by the reach of the adjacent gradient of human influence on non-altered 'wild' or natural land where ecological functions, such as the natural disturbance regime are affected. As a consequence of the change in the natural disturbance regime there is a change in ecological processes such as successional pathways or ecological services such as carbon sequestration. The EWUI is an expansion of the traditional human population/vegetation density based WUI often used for fire safety studies (Radeloff et al. 2005; Zhang et al. 2008) and is in line with the idea of the ecological affect area (Leu et al. 2008) and the boundary framework (Cadenasso et al. 2003), but focuses specifically on human caused disturbance. Additionally, the EWUI specifies the area affected and the rate and intensity of ecological change. Defining the EWUI requires several steps:

1. A measure of human influence adjacent to the natural area (e.g. altered land)
2. A record of spatial/temporal disturbance regime (e.g. fire)
3. A spatial measure of an ecological process (e.g. changes in forest composition)

4. A method of evaluating the spatial extent of the gradient of human influence on: a) the disturbance regime and b) the ecological pathway (e.g. buffers of altered land).

There are several scales to consider when defining the EWUI, the human influence scale, the abiotic process scale, and the biotic process scale; therefore it is essential that all perspectives are considered when attempting to define the scale of the EWUI. Using the EWUI framework, we can begin to understand how altered landscapes indirectly affect disturbance regimes and ultimately, the range and extent of the ecological consequences of those changes. To illustrate the process of obtaining the inputs for analysis and defining a specific EWUI for an established natural area, we used fire frequency and forest composition change in relation to proximity to altered land in the Pinelands of New Jersey.

Background: New Jersey Pinelands National Reserve (NJ PNR)

Management challenges

The New Jersey Pinelands National Reserve (NJ PNR) is an International Biosphere Reserve and a fire-dependent ecosystem, located in the most densely populated state in the USA (Figure 1). It has been highly influenced by human interactions from pre-settlement to today (Wacker 1998). The Pinelands has experienced high development pressure given its close proximity to New York City and Philadelphia as well as the popular Jersey Shore beach destinations. In 1978, the creation of the National Reserve formed a core natural area surrounded and interspersed with varying degrees of development and LULC intensity; with zoning and development levels enforced by a regional land use planning entity, the Pinelands Commission. A recent review of LULC trends in New Jersey indicated that the Pinelands National Reserve area has been highly successful in limiting human development in designated preservation zones there is still has up to 25% of land available for development within designated development zones (Hasse and Lathrop 2010). Although the enhanced risk of wildland fires to human life and property within the WUI has received much attention by Pinelands managers; other more immediate concerns such as fighting fires, regulating development, and protecting endangered species has prevented local land management agencies from focusing on the potential impact of long term changes to natural disturbance regimes as a driver of forest composition change.

Though there are extensive areas of contiguous forest land, a ‘let burn’ policy is difficult in the NJ PNR due to the close proximity of residential and business areas to fire-prone forests. Suppression efforts utilizing air and ground support during the latter part of the 19th century and early 20th century have been highly effective compared to historic accounts and the mandate of the New Jersey Forest Fire Service (NJFFS), created in 1906, has been immediate suppression for safety reasons (Barresi 2002). Within the conservation community, there is widespread concern that active wildfire suppression, as well as prescribed fire regimes, may be affecting long-term succession and vegetation community composition in the Pinelands, creating a shift from a pine to an oak-dominated forest system. A significant challenge in the Pinelands is to incorporate these community and ecosystem level interactions into long term management goals of maintaining fire disturbance for fuel reduction and maintenance of the pinelands ecosystem, while also protecting human health, life and infrastructure. The ultimate success of the Pinelands National Reserve in maintaining this internationally significant ecosystem and its numerous endemic fire-dependent plant species will depend on our ability to establish ecologically sound fire and land use management policies.

Fire history and ecology

Contemporary published fire histories compiled for the Pinelands are limited to a PhD dissertation study (Windisch 1999) and Buchholz and Zampella's (1987) 30 year fire history, both of which focused on the unique dwarf pine plains region. Forman and Boerner's (1981) study of fire statistics aggregated fire data across the entire southern region of the state from 1914 to 1975 (from Divisions B and C, Figure 1). While these investigators compiled and examined long term fire histories for the Pinelands, they were limited in spatial extent and/or lacked modern computer mapping and advances.

Boerner (1981) concluded that wildfire kills most oak species in the Pinelands of NJ, but only reduces biomass for *Pinus rigida* P. Mill. (pitch pine), the dominant pine species. *P. rigida* is remarkably adept at surviving fire with its thick bark. With less severe fires, *P. rigida* resprouts epicormically from the bole of the tree as well as the limbs creating a "furry tree" type appearance. More severe crown fires tend to kill the crown of the tree but promote basal resprouting (Forman and Boerner 1981). This species is also known for its cones which can be serotinous, releasing seeds only after fire (Good et al. 1998). Smaller oaks such as *Quercus marilandica* Münchh. (blackjack oak) and *Quercus ilicifolia* Wagenh. (scrub oak) as well as the *Quercus stellata* Wagenh. (post oak) are also excellent reprouters after fire; however, many of the larger oaks such as *Quercus velutina* Lam. (black oak), *Quercus alba* L. (white oak), and *Quercus coccinea* Münchh. (scarlet oak), may not survive intense fire (Collins et al. 1994). In addition, Little (1998) noted that in the absence of fire, pine-dominated areas succeed to oak-dominated forests. An area of the NJ Pinelands termed the pine plains is known to have higher fire frequencies (Windisch 1986), exhibiting pines of shorter stature and a higher abundance of serotinous cones (Givnish 1981).

Pine versus oak: Advantages and disadvantages

The distribution and ratio of pine versus oak cover is directly related to fire disturbance and an understanding of historic and more recent distributions across the landscape is essential in creating management goals. Historic accounts of the ratio of pine to oak cover in the Pinelands are limited to general descriptions or maps of the landscape (Harshberger 1900; Stone 1911; Vermeule 1889). A decline in oak pollen versus pine pollen was experienced in the region of the Cape Cod, Massachussetts pinelands after colonial settlement due to increased human caused wildfire (Parshall et al. 2003), but there is no pollen research specifically for the New Jersey Pinelands investigating how forest composition may have changed when settlers increased or decreased native disturbances. Pine versus oak forest composition of the Pinelands was first quantitatively inventoried with Landsat satellite images from 1972 to 1988 (Luque et al. 1994). This study showed that the fringe areas of the Pinelands were changing in forest composition with forest succeeding from pine-dominated to oak/pine dominated forests (Luque et al. 1994), presumably due to modern fire suppression efforts.

One advantage of pine to oak shifts in WUI areas, with less fire-resilient species (such as *Quercus alba* L.), is that oaks may lessen fire severity due to the higher water holding capacity of oak litter and green broadleaf leaves. They also have less undergrowth or ladder fuels due to greater leaf areas and shading capacity (Horace Somes, Division B supervisor retired and Berte Plante, Division B supervisor, NJFFS, personal communication). However, in addition to the loss of endemic and endangered species habitat, disadvantages of such a shift include a greater susceptibility to gypsy moth and storm damage for oaks which may dampen the ability of oaks to serve as fire breaks and instead may provide standing, dry, deadwood fuel for those areas affected for several consecutive years. In addition, off-road fire-fighting trucks are typically used to drive over small pines to access and suppress life and property threatening fires. It is more difficult to access fire in oak dominated canopies compared to pine dominated canopies, a major safety concern for wildland firefighters (Berte Plante, Division B supervisor, NJFFS, personal communication).

Using the study area of the Pinelands of New Jersey in defining the EWUI concept will help managers quantify how altered land influences the adjacent fire regime and forest composition through space and time. The Pinelands study area provides an island of natural forest surrounded by a wildland urban interface that is likely to typify areas with high development pressures and provides an example of the effects of human-environment interactions on ecosystem function.

Methods

Study Area

The Pinelands of New Jersey (Figure 1) contain generally highly porous, acidic, sandy soils of the Cohansey formation, created from ocean floor deposits around 5 million years ago (Rhodehamel 1988). These sandy soils contribute to the low water retention in this area. Segregating the area and focusing on watershed level fire and LULC history rather than the NJ PNR as a whole is critical to establishing baseline data for concurrent research on the relationships between LULC, fire, and water quality measurements

(Chapter 2). In addition, the Barnegat Bay watershed is approximately seven times more populous than the Mullica River watershed providing a template for possible future trends for the EWUI in the Mullica River (Lathrop et al. 2003; Lathrop and Conway 2001). The Barnegat Bay and Mullica River watersheds within the NJ PNR represent a large part of the unique ecosystem and landscape (Figure 1) that reflects a long history of fire disturbance, largely anthropogenic (lightning accounts for ~1% of large Pinelands fires over the recorded fire history (Barresi 2002)) . I used these watersheds as case studies investigating gross changes in altered land and fire regimes as well as differences in forest composition to determine the EWUI.

Human influence (defining the EWUI, requirement1)

Defining the extent of the EWUI required a comprehensive review of the land use land cover datasets of New Jersey in order to understand the extent and dynamics of altered land in the Pinelands. To create an altered land category for Barnegat Bay and Mullica River watersheds, altered land was selected from NJDEP LULC of 1986 delineations based on color infrared (CIR) stereo-paired hard copy imagery, dated 1986 and with an accuracy update of 1986 LULC boundaries using USGS digital CIR orthophotos dated 1995 and 1997 (NJDEP 2000) and 2002 (LULC delineations based on 2002 digital CIR orthophotos) which included LULC level 1 agriculture, barren and urban cover types as well as level 3 managed wetlands and cranberry farms (see Table 1, NJDEP 2006). In addition, LULC level 3 classes comprising the altered land category included wetland right-of-ways, managed wetlands in maintained lawn, managed wetlands in recreation areas, and agricultural wetlands under the wetlands level 1 category. Altered land polygons for each year were converted to raster format with a 30m pixel resolution. The

30m pixel size and watershed boundaries facilitated later modeling efforts and were consistent throughout our analysis.

The focus of altered land change was between 1986 and 2002; however, as validation for our study and to gain understanding of EWUI trends over time, the change in altered land was also evaluated for 2002 and 2007 LULC dates. The 2007 LULC update incorporated new water features and reclassified transportation corridors based on 2007 CIR orthophotos. These features were retroactively mapped to 2002, but the LULC classifications did not otherwise change from the original 2002 release (Table 1, NJDEP 2006; NJDEP 2010). The validation portion of our analysis was performed on the change between 2002 and 2007 and not in combination with earlier dates.

Disturbance regime (defining the EWUI, requirement 2)

Understanding the history of fire and the disturbance regime is the second step in defining the EWUI of a particular landscape; therefore a significant part of the project reported herein included compiling a digital fire history geodatabase for landscape analyses by digitizing paper map illustrations of large fire perimeters. The fire geodatabase contains a dataset consisting of all of the recorded wildfire perimeters (i.e., outer bounds of the fire event footprint) considered large fires (> 40.47 ha, 100 ac) in the archives of the NJFFS from 1924-2007 (Figure 1). Wildfires are defined here as ignited by humans or by natural causes but not prescribed. Large wildfire records were limited for the years 1975-1978 due to the loss of paper maps associated with fire records during these years. Perimeters were verified and edited using 1930's aerial photography mosaic of New Jersey (NJDEP) for fires occurring prior to more recent aerial photography. Recent aerial photography used for reference in locating fire perimeters included hard

copy mosaic airphotos from 1956 and 1963 (USDA 1956; USDA 1963) and digital orthophotos from 1995/1997 and 2006. In addition, a digital mosaic of USGS topographic maps ranging in dates from 1954 to 1997 was used to verify locations and landmarks. All efforts were made to digitize fire perimeters as precisely as possible however, accuracy depended upon several factors which varied depending on the following resources: the background imagery, the detail of the map on the NJFFS paper fire record (some were quite detailed, others were merely outlines with latitude and longitude), and the digitizing itself. Perimeters may not always have excluded areas within the fire event boundary (i.e., interior "islands") that did not burn (e.g., wetlands); therefore the total area burned may be less than recorded for some fire perimeters. However, obvious, large water bodies were excluded when digitizing large wildfires if imagery dates and fire dates were similar. More recent fire perimeters can be considered more accurate due to use of GPS technology (2004-2007) on the part of NJFFS.
All spatial sources were cross-referenced to gain the best match of fire perimeters to available landmarks. Fires for which no fire records exist could not be included; therefore the large wildfire dataset should be considered a complete dataset for which there are verifiable paper fire map records and no more. Additionally, fire severity was not reported in paper fire records and could not be utilized in this study. The Barnegat Bay and Mullica River watersheds experienced 516 large wildfire perimeters (based on hectares calculated) in the geodatabase.

Prescribed fire areas were excluded in the analysis for all stages of our methods. This distinction was important in ensuring that the entire history of fire frequency calculations (1924-2002 or 1924-2007) was not confounded by more recent overlapping prescribed fire occurrences. Prescribed fire is performed repeatedly on limited areas of the Pinelands (Figure 1) and the reliability of the prescribed fire history is severely limited compared to the wildfire records. Additionally, low severity winter and early spring prescribed fire is used mainly for fire breaks, fuel reduction and the creation of wildlife feeding areas and results in the maintenance of canopy species.

The number of fires at all locations across the landscape were computed by converting all perimeters per year to raster format with a 30m pixel size and summing overlapping fire years (from 1924-2002 for the main study and 1924-2007 for our validation study) in ERDAS Imagine (ERDAS Inc. 2009). There were no known areas on the landscape that burned more than once in a particular year. Also evaluated were hectares burned per year across the study area.

An ignition point feature class was also created using all available ignition points (associated with a fire >0 ha) that occurred in the two study watersheds. The ignition points with latitude and longitude information obtained from the in-house NJFFS Form 2 fire record database from the years 1929, 1936-37, 1939-44, and 1991-2006 were included in this analysis based on availability of data alone. The ignition points spatial database contains only those fires that also included location information and fire size for the associated fire record and should therefore be considered a limited sample of total ignitions recorded by NJFFS. These records do not include prescribed fires. Of the total number of ignition points in the database for the entire New Jersey area (33740 as of 2007), 7287 were found to have recorded latitude and longitude and a size greater than zero falling within the Barnegat Bay and Mullica River watersheds. Most ignitions were triangulated from fire towers and were recorded to the nearest 15 seconds (of latitude and longitude) thus producing a square pattern in some areas (Horace Somes, Division B Supervisor, NJFFS retired, personal communication). Ignition data were extracted using the intersection tool in ArcGIS 9.3 (Redlands, CA) and analyses included densities for each buffer transition area category (see methods below).

Ecological pathway (defining the EWUI, requirement 3)

To enumerate forest composition change, six generalized forest classes were condensed from LULC Level 3 classifications and converted to raster format with a 30m pixel size: upland deciduous (UPDEC), upland coniferous (UPCON), upland mixed (UPMIX), wetland deciduous (WETDEC), wetland coniferous (WETCON) and Atlantic white cedar (ATWHCED) (Table 1). Areas that were classified as burned, shrubland, oldfields, dunes, plantations, saline marsh, herbaceous wetlands or phragmites level 3 categories for either date were excluded from evaluation of forest successional trends. Additionally, the former agricultural wetland/becoming shrubby/not built up category was excluded since the forest composition (for either 1986 or 2002) of these areas was non-existent or unknown and the focus of the EWUI analysis was on changes in forest composition rather than succession from non-forest to forest.

Pontius (2004) developed a method of evaluating land cover change to differentiate observed vs. expected LULC change results, where expected change is viewed as swapping of equal areas at different locations of LULC types and/or persistence of current LULC categories. Using the Pontius method enabled us to identify areas of systematic change, or the loss of one forest category contributing directly to the gain of another in the same location. In our study, any forest category that departed from the distribution of the relative proportion of forest classes at time 1 was attributed to change above and beyond persistence or swapping of equal areas. This type of change has been termed systematic change (Pontius et al. 2004). Identifying systematic change gives us the ability to enumerate areas of forest succession as opposed to unimportant random change. We adopted this method to examine upland forest change between 1986 and 2002. Expected loss distributions are based on the following equation:

$$L_{ij} = (P_{i+} - P_{ii})\left(\frac{P_{+j}}{\sum_{j=1, j\neq i}^{J} P_{+j}}\right) \quad (1)$$

Equation 1 depicts expected loss from a forest category in 1986 to another forest category in 2002 (L_{ij}) as a product of the total amount of land in the category of interest for 1986 (P_{i+}, the row total in Table 2) less the area that persisted as that same category from 1986 to 2002 (P_{ii}, the diagonal in Table 2) multiplied by the ratio of total amount of land in that category in 2002 (P_{+j}, the column total in Table 2) to the proportion of land in 2002 not accounted for by persistence in the category (P_{+j}, $j \neq i$, the column total excluding persistence from the diagonal). In essence, in the case of the expected loss from coniferous to deciduous forest (UPCON to UPDEC), the equation takes the loss in coniferous forest from 1986 to 2002 and distributes it across all non-coniferous forest categories in 2002 while also accounting for the relative size of the deciduous category in 2002. These transitions were evaluated in terms of both gain and loss due to the potential shifting to any of 6 categories of forest cover categories. The expected loss distributions calculated in Equation 1 were altered to reflect expected gain. Row values were replaced by column values and vice versa. In the case of gain (when one forest category comes ‘from’ another category) in 2002, in comparison to the 1986 distribution, we can pinpoint the classes that disproportionately contributed to gains in forest types for 2002. Alternately, in the case of loss from 1986 (when one forest category goes ‘to’ another category), we can pinpoint whether the ranking of forest categories departed from expected distributions when transitioning to 2002 categories. If the gain in a category such as upland deciduous forest (UPDEC) is coming foremost (e.g. observed minus expected percentage ranks highest in this category) from upland coniferous (UPCON) and the loss in UPCON is going foremost to the UPDEC category we can conclude that the transition is systematic (Pontius et al. 2004). If the gain in UPDEC is coming foremost from UPCON, but UPCON is transitioning primarily to UPMIX, the transition from UPCON to UPDEC is not considered systematic because the majority of the loss in UPCON is going to UPMIX, with a lower percentage transitioning to the UPDEC category.

Forest transitions included 36 possibilities (6x6 forest categories) all of which were evaluated for systematic change within specified areas of study (see below). Additionally, change in total hectares of systematically changing forest classes were evaluated between 1986 and 2002.

The spatial extent of human influence (defining the EWUI, requirement 4)

The scale of the EWUI gradient in this study was defined using a buffer distance analysis (i.e. distances from the altered land into the forest), essentially categorized or binned into a series of parallel buffer zones at certain distances from altered land. In order to create a measurable ecological wildland urban interface around altered land, we used the search function to create a distance from altered land layer in ERDAS Imagine (ERDAS Inc.

2009) and binned the results into six buffer distance areas extending from the edge of the altered land into the forested areas of the Pinelands. A seventh category encompassed everything beyond the final buffer and represents the 'interior' forested areas of the Pinelands for each watershed. Buffer widths were graduated from a width of 120m adjacent to altered land (0-120m) to a width of 600m for the last buffer (from 1680-2280m) in order to better display the extent of human influence in areas closer to altered land. All buffer widths are divisible by 30 because of the 30m pixel resolution of all of our raster layers.

Buffer distances from altered land changed from 1986 to 2002 due to landscape configuration differences associated with new development as well as forest growth (Figure 2). Thus a matrix of 49 buffer transition areas (BTA) was created when seven buffer categories from 1986 were intersected with seven buffer categories from 2002. This matrix was used as the basis for specifying the relationship of any point within the forest to altered land for 1986 as well as 2002. If altered land expanded or 'leapfrogged' into a forest area, an area of forest may have been considered farther from altered land in 1986 and closer to altered land in 2002 (i.e. transitioning to a buffer distance category closer to altered land). If an area was classified as altered in 1986, but through natural vegetation succession was classed as old field in 2002, a change in the altered land buffer configuration for 2002 resulted and an area of forest shifted from a buffer category closer to altered land in 1986 to a buffer category farther from altered land in 2002 (i.e. transitioning to a buffer distance category further away from altered land). The categories of 'transitioning closer' and 'transitioning away' from altered land are referring to *temporal differences in classification* between the years 1986 and 2002, not movement in *location on the landscape*.

Our analysis focused on buffer transition areas accumulating 100 ha or more per category. Smaller BTA categories were eliminated since they comprised such a small percentage of the study area. In addition, it was found that systematic forest composition transitions rarely occurred when the cumulative BTA area was less than 100 ha. Once systematic changes in forest cover in each of BTA category greater than 100 ha were evaluated, the difference in area in hectares of gain or loss was used to depict change in each forest category. Understanding whether shifts in UPCON vs. UPDEC occurred was the focus of this analysis with other forest categories included in order to account for transitions to all possible forest types.

Additionally, to assess the relationship between fire and forest composition, histograms of the percent of each frequency category in each systematic forest change category were created to understand the contribution of fire frequency to these changes. Fire frequency was also summarized for each BTA in its entirety to elucidate the relationship of fire regimes to altered land. Due to the numerous combinations of BTAs (49), 5 representative BTA combinations from 1986 and 2002 were investigated for fire frequency: 0-120m to 0-120m (remaining **adjacent to altered**, no transition), 480-1080m to 0-120m (**transitioning closer** to altered), 480-1080m to 480-1080m (remaining midway between altered land and interior areas ; **middle**), and 2280+ m to 2280+ m (remaining **interior natural area,** no transition) (Table 3). These measures were also calculated as a percentage of the entire watershed (excluding water bodies) to elucidate the magnitude of landscape level effects. Ignition densities were extracted for each BTA to understand the relationship between the number of ignitions and the actual area burned as well as the proximity to altered land and the effectiveness of historic fire control measures.

The forest composition change categories derived from the dates 1986 to 2002 represent two points in a continuum of forest disturbance and succession. However, we used the full fire frequency history available (1924-2002) in comparison to recent forest composition change to enhance our understanding of how fire frequency in different locations contributes to these more recent change trajectories.

Investigating trends with the 2007 LULC update

All of the methods above were repeated for the 2002 to 2007 LULC years (fire years 1924-2007) for validation purposes and to elucidate any continuing trends.

Results

Human influence (defining the EWUI, requirement1)

Changes in forest proximity to altered land show that most of the Pinelands forest remains in a similar proximity to altered land from 1986 to 2002 (Table 3 diagonals). Even though the buffer distance area remaining closest to altered land was one of the least broad (120m), it contained the most forested land in Barnegat Bay. Mullica River contained the most land in the area remaining 480m-1080m from altered land. For both watersheds the buffer transition areas beginning at 120m-240m in 1986 and ending at 0-120m in 2002 contained some of the highest amounts of land (3250.17 ha and 2156.13 ha for Barnegat Bay and Mullica River watersheds respectively). Only 383.67 ha in Barnegat Bay and 3463.11 ha in the Mullica River watershed were considered interior natural areas according to our definitions for both 1986 and 2002 (greater than 2280m from altered land).

Disturbance regime (defining the EWUI, requirement 2)

The frequency of large (i.e., > 40.47 ha, 100 ac) wildfires during the 1924-2002 time period across Barnegat Bay and Mullica River watersheds depict an extensive fire prone ecosystem and the pine plains generally have a higher fire frequency than the rest of the Pinelands landscape (Figure 3). In 1963, after a severe drought, up to 82,000 ha of the central Pinelands burned in a wildfire, that began on April 20th (Forman and Boerner 1981). The number of hectares burned has decreased through the time period of record, with a marked decrease after the 1963 fire season (Figure 4). The average number of hectares burned before 1963 was 4586ha/yr; while the average after 1963 has been reduced to 1987ha/yr (calculations do not include 1963). In addition, there are six years with no large fires after 1963 (not including the years 1975-1978 – see Methods); whereas 1924-1962 had no recorded years without large fires.

Ecological pathway (defining the EWUI, requirement 3)

Overall, percent change in upland forest types was evident within the watersheds over the 16 year time period of study (Figure 5); with Barnegat Bay showing the most dramatic change in upland deciduous forest cover (9% relative gain between 1986 and 2002). These overall changes depict the relative distributions through time between upland forest types; however, our focus was on using the Pontius (2004) method to understand whether the change in forest composition was 'systematic' in areas closer to altered land vs. farther away. Systematic changes were only found within the UPDEC, UPCON and UPMIX and were the focus of the EWUI analyses. Barnegat Bay UPCON lost approximately 900 ha (~2200ac) to UPDEC cover during this period and Mullica River UPMIX lost approximately 200 ha (~500ac) to UPDEC.

Systematic change was evaluated and percent change (Table 4) as well as hectares were calculated (Table 5) for UPDEC, UPCON and UPMIX categories. Each of the forty-nine buffer transition areas was evaluated, with adjacent to altered, transitioning away from altered, transitioning closer to altered, middle, and interior natural areas highlighted in this study (see Methods for explanations of these buffer distance categories, Tables 2 and 3). In all cases and for both watersheds, transitions from UPCON to UPDEC were systematic in areas closer to altered land (e.g. UPCON lost primarily to UPDEC and UPDEC gained primarily from UPCON, Table 5). In addition, in areas where UPMIX was lost, the loss was systematic to UPCON except in the BTA closest to altered land in which case UPMIX lost systematically to UPDEC gains. This resulted in a forest succeeding dichotomously toward UPCON in the interior, or UPDEC in the fringe areas. UPDEC either gained hectares or remained the same across the entirety of Barnegat Bay, whereas UPCON lost in the 0-480m range and gained slightly in the 480-2280+m range. The largest changes were in those areas that have been the closest to altered land for the longest time period (e.g. 0-120m in 1986 as well as 0-120m in 2002). The Mullica River watershed showed similar trends in the areas closest to altered land with transitions from UPCON to UPDEC; but UPMIX losses to UPDEC had a larger influence in this watershed closest to altered land. In addition, some middle to interior areas transitioned systematically from UPMIX to UPCON rather than to deciduous forest cover in the Mullica River watershed.

The spatial extent of human influence (defining the EWUI, requirement 4)

Within each forest composition change category we see a difference of relative area experiencing specific fire frequency categories. For Barnegat Bay, forty-one percent of areas remaining deciduous from 1986 to 2002 have not experienced a large fire over the 78 year fire record (Figure 6a). Approximately 27% of areas moving from UPCON to UPDEC have not burned during this period illustrating a time when fire conditions were more conducive to the 1986 UPCON conditions. Eighty-four percent of the areas staying coniferous have burned more than once during 1924-2002. In addition, 93% of the areas moving from UPMIX to UPCON burned at least once during the period of record.

For the Mullica River watershed, we find a more muted signal of the effect of fire frequency on forest composition changes, although similar trends appear with 80% of the UPMIX to UPCON category exhibiting one or more burns (Figure 7a). The differences between the fire frequency distributions are not as distinct as in Barnegat Bay although UPCON to UPCON and UPMIX to UPCON are skewed toward higher fire frequencies and UPDEC to UPDEC and UPCON to UPDEC are skewed toward lower fire frequencies tracking the trends in Barnegat Bay. Overall, the total area of upland forest composition change categories investigated (UPDEC to UPDEC, UPCON to UPDEC, UPCON to UPCON and UPMIX to UPCON) make up a similar proportion of the watershed in Barnegat Bay (28%) compared to the Mullica River basin (33%).

Fire frequency across different BTAs (i.e. different distances from altered land) of Barnegat Bay display very different distributions in relation to altered land; with 44% of areas closest to altered land having no large fire in the last 78 years (Figure 6b). Interior natural areas are highly susceptible to fire with 89% of interior areas burning 2 or more times in the fire history. The adjacent to altered land (0-120 m) forested category in Barnegat Bay makes up 28% of the entire watershed and the interior category (greater than 2280 m) makes up only 0.5% of the total watershed area.

The Mullica River watershed shows similar fire frequency patterns compared to Barnegat Bay for buffer transition areas, but with fewer fires overall, skewing all distributions towards lower fire frequencies (Figure 7b). Middle areas and interior areas are slightly skewed toward more fire frequencies when compared to other BTAs within the Mullica River, but again, the differences between frequencies are muted as compared to Barnegat Bay. Interior areas show the largest percentage of area burning one time in the fire history, but middle areas actually demonstrate higher percentages in the two or more times burned categories than interior areas. Much of the Mullica River watershed BTAs displayed fall within the category closest to altered land as well as the middle BTA, although the interior areas are apparent.

Ignition densities for Barnegat Bay and Mullica River watershed show that areas closest to altered land have the most ignitions per hectare (Figure 8) and Barnegat Bay has higher ignitions per hectare than Mullica River overall. Other BTA areas show that ignitions are relatively rare in comparison with those areas remaining closest to altered land.

Defining the EWUI in the Pinelands of New Jersey

The rate of ecological change and spatial extent of the ecological wildland urban interface (i.e. the influence of altered land on the fire regime and pine to oak succession in adjacent protected forest), can thus be specified in the Pinelands by examining fire frequencies and forest composition changes within buffer transition areas. The extent, or reach of the EWUI into natural forested areas in this case can be recognized by the BTAs where coniferous or mixed forest are systematically losing hectares to deciduous forest gains and where these transitions can be linked to fire frequency. If fire frequency is low near altered land as well as in areas that transition from pine to oak, we can assume that areas closest to altered land either have, or will exhibit this type of forest transition. Areas in Barnegat Bay in the UPDEC to UPDEC category show a fire frequency pattern similar to the adjacent to altered BTA category (Figures 6a and 6b). Areas in the UPCON to UPDEC category show a fire frequency pattern similar to the area transitioning closer to altered land. UPCON to UPCON areas show a similar distribution to the middle category and interior areas generally show a higer percentage of the interior area falling within the two times burned categories, similar to the UPMIX to UPCON forest transition fire frequencies.

Comparing forest composition change fire frequency distributions to BTA fire frequencies in the Mullica River watershed, we see that the middle BTA is similar to the UPCON to UPCON forest composition change frequency and that UPDEC to UPDEC and UPMIX to UPDEC are similar to the adjacent to altered BTA category. Upon further investigation into the locations of high UPMIX to UPCON fire frequency distributions it was found that the areas in the 1080-1680m and 1680-2280m BTAs (results not shown) were similar to the UMIX to UPCON. This corresponds to the loss of UPMIX and gain of UPCON in these areas from 1986 to 2002 (Table 5, Mullica River watershed).

Based on our EWUI definition and on the buffer distances evaluated, the influence of the proximity of altered land on upland forest composition in these watersheds extends from the edge of altered land up to 480m into adjacent upland forests in Barnegat Bay and up to 240m in the Mullica River watershed (Table 5). Upland forests within the EWUI buffer areas comprise approximately 19% of the Barnegat Bay watershed and 11% of the Mullica River watershed.

Investigating trends with the 2007 LULC update

All methods were repeated using 2002 to 2007 LULC and results showed that altered land differences, fire frequency and forest composition change was minimal between these two dates. Additional fires were not relevant in that recently burned areas were removed from analysis since a 'from/to' forest composition could not be determined in burned areas. Systematic changes in forest composition change were not apparent in the BTAs for these dates; however, overall trends in upland forest change show that the Barnegat Bay and Mullica River watersheds are continuing to lose upland coniferous forest and gain upland deciduous forest at the landscape scale (Figure 5).

Discussion

The first steps of defining the EWUI

Understanding how altered land has changed through time and space was the first step in defining the EWUI. The amount and location of altered land influenced the magnitude of the buffer transition areas between 1986 and 2002 (Table 3). Both watersheds show large segments of land shifting closer in proximity to altered land from the 120-240m distance category to the 0-120m distance category. This shift depicts an area with expanding altered land near the edges of already developed land. Interestingly, areas transitioning away from altered land (above the diagonal) were on par with areas transitioning toward altered land (below the diagonal) at the entire watershed scale (Table 3). This is not surprising given the implementation of development restrictions in the NJ PNR along with the potential for forest regeneration from old fields and abandoned agricultural areas. The 2007 update showed even less forested areas transitioning toward and away from altered land confirming some stabilization of the influence of the proximity of altered land in the NJ PNR on adjacent forested land. The fact that the Barnegat Bay watershed contained only 383.67 ha of forested land greater than 2.28 km from altered land during the entire 16 year period shows that at the watershed scale, the interior forest category is only a small fragment of land. The Mullica River's interior forest status was better with 3463.11 ha more that 2.28 km from altered land for the entire 16 year study period. These areas should be targeted for enhanced protection for those species and ecological processes requiring large areas of contiguous forest away from development.

Compiling the spatial fire history of disturbance for the Pinelands was the next step in defining the EWUI for the Pinelands. The Pinelands have an extensive fire history and the unique dwarf pine plains area has more frequent fire than the rest of the Pinelands landscape (Figure 3, data not shown). The fire regime in the Pinelands has been highly influenced by humans via suppression. The steady decline in fire occurrences as well as total hectares burned per year shows the effectiveness of the suppression policy. The difference of fire size before and after 1963 is clear in the current dataset (Figure 4). The initial drop in hectares burned per year after the 1963 fire was most likely initially due to the decrease in available fuels; however, the events of 1963 also spurred the focus on equipment (via excess Federal war equipment) and technology within the NJFFS increasing access to fires for wildland fire fighters (Bert Plante, Division B supervisor, NJFFS, personal communication). Other factors for the reduced annual hectares burned in the later part of the century could include increased use of prescribed fire (Barresi 2002), an increase in human population concerned with suppression near their homes along the coast of New Jersey, and the more dispersed towns in the Mullica River watershed (Lathrop et al. 2003; Lathrop and Conway 2001). All of these factors likely played a part in the decrease in the average fire size after the 1963 fire year.

Comparing this fire history to other fire history compilations in New Jersey is difficult due to the difference of scope. Forman and Boerner (1981) describe data from 1914-1975 that classifies the NJFFS Division B and C as Pine Barrens. In actuality these Divisions incorporate an area that ranges from the tip of the Cape May peninsula north to the Raritan River; a much larger area than the Barnegat Bay and Mullica River watersheds or the Pinelands National Reserve alone. Accordingly, their records seem to reflect higher average yearly hectares burned and a split around 1940 with higher hectares per year before 1940 and lower hectares per year after 1940. This split is not readily apparent in our study area.

The mean fire return interval in Barnegat Bay is shorter by 32 years than that of Mullica River (77 and 109 years respectively). This indicates that large fires are more common in Barnegat Bay and that disturbance may be more of a defining factor in forest composition change in this area.

The spatial extent of human influence

Forest composition changes from UPCON and UPMIX to UPDEC are the most interesting in this study because they indicate successional trends away from the Pinelands ecosystem for which this area was protected. The largest impact of these changes is in areas that have been closest to altered land for the longest time period (0-

120m for 1986 and 2002). These figures indicate that the areas closest to altered land are at risk of continuing to succeed toward UPDEC cover in the future given a continuation of the current altered land and fire regime. The Mullica River watershed shows a dichotomous change with areas closest to altered land moving from UPMIX to UPDEC and areas further away from altered land moving from UPMIX to UPCON. Further investigation into these phenomena is warranted in order to understand the underlying forces behind differing trajectories. In the case of UPMIX to UPCON, fire frequencies leading to this transition could be utilized to return areas of UPMIX to UPCON cover where desired for management purposes.

The clear distinction between forest composition change categories for relative fire frequencies in the Barnegat Bay watershed follow known ecological pathways in the Pinelands (Figure 6a), with the UPMIX to UPCON transition requiring more frequent fires to occur than the UPCON to UPDEC transition, which requires little to no fire to occur (Little 1998). The effect overall in the Barnegat Bay watershed for areas transitioning from UPCON to UPDEC is relatively small with only 4% of the watershed experiencing this transition, although where these transitions are occurring (next to altered land), is noteworthy. However, with 19% of the watershed classified as upland forest in the EWUI, the change could be readily apparent to the public. This study demonstrates that areas of the EWUI not experiencing fire could become UPDEC within the next 100 years if the rate of change continues on a similar track.

Mullica River watershed forest composition transitions are also related to fire frequency, although differences in fire frequency distributions are not as striking as in Barnegat Bay. The question becomes, then, what is regulating forest composition change in the Mullica River basin if fire frequency is playing a lesser role? Why does the UPCON area not generally need to be burned more than two times to stay UPCON (as we see in Barnegat Bay)? What is slowing down the rate of succession from coniferous to deciduous cover in the Mullica River? Perhaps the soil distribution in the Mullica River basin is more conducive to UPCON persistence in those areas where it does exist; or more of the Mullica River area burned later in the fire history than Barnegat Bay; therefore the trajectories of forest succession due to fire frequency may not be as apparent from 1986 to 2002, but may become more apparent over time. An increase in coniferous and understory biomass would be the dominant process under more recent fire rather than succession to a different forest type. These hypotheses should be investigated further in order to understand the drivers of forest composition change in the Mullica River watershed. Only about 1% of the entire Mullica River watershed is transitioning from UPCON to UPDEC, although again, these areas tend to be closest to altered land. Approximately 11% of the Mullica River watershed is upland forest in the EWUI area. This means that if the UPCON to UPDEC continues at the same rate it will take less than 200 years for the upland forest in the EWUI of the Mullica River to transition to UPDEC without fire. The percent watershed figures illustrate why it is difficult to discern how fire frequencies affect forest composition changes when looking at the percentage of forest categories within the entirety of the watershed. Other important yet small shifts in forest cover categories can be overlooked when focusing on the region as a whole.

Making the EWUI connection: proximity, disturbance and composition

Finally, the connections between proximity to altered land, differences in the disturbance regime and the resulting changes in forest composition are apparent. Statistically, each buffer transition area or forest composition area represents a sample of one; therefore the skewed left or right fire frequency distribution of the BTA sample versus the forest composition transition sample cannot be quantitatively compared in this type of analysis. However, at the landscape level, it is clear those areas closer to altered land over the period of study exhibit disturbance regimes conducive to upland deciduous transitions or persistence. The relative distribution of fire frequencies in areas adjacent to altered land mimic the fire frequencies seen in the UPDEC to UPDEC forest composition category for Barnegat Bay. The distributions in the middle BTA (480-1080m for 1986 and 2002) mimic the UPCON to UPCON category. Areas transitioning away from and transitioning closer to altered land have a larger percentage of zero and one time burned fire frequencies than areas staying in the middle or interior categories. The fact that areas shifting proximities from middle to closer to altered land are trending toward closest to altered land fire frequencies could be a lag effect in that the fire frequency is based on 78 years, whereas the altered land buffers intersections were based only a 16 year transition period (1986-2002). For example, an area that was farther from altered land in 1986 most likely experienced more frequent fires before 1986 or until the point at which adjacent areas were developed. Fewer fires in the Mullica River watershed history could explain the dampened effect of altered land on forest composition, but the timing of fires in the fire history should not change the effect of overall fire frequency as it relates to altered land. Although we considered agriculture as altered land, a greater percentage of agriculture in the Mullica River watershed may dampen the effect of altered land due to more open space and less residential fire suppression needs in agricultural areas.

It is interesting to note that ignitions in each watershed are higher near altered land (associated with higher human populations) and taper off in the interior natural areas. Although this type of pattern of ignitions close to human population is common across numerous fire prone systems (Catry et al. 2009; Romero-Calcerrada et al. 2008; Syphard et al. 2007), this means that ignition point densities for fires of any size in the Pinelands (Figure 8) have an inverse relationship to large fire frequency (Figures 6 and 7) as we move across the EWUI gradient towards interior areas. Highly effective fire suppression near life and property and perhaps the less volatile nature of deciduous cover in these areas are possible explanations for this inverse relationship between ignition density and large fire frequency. Whether a large fire which ignites in the EWUI typically spreads toward interior forest or stays in the EWUI cannot be determined from this study due to the difference in dates for determining the extent of the EWUI area (1986 and 2002) and the dates of historic fires (1924-2002). However, studies which use ignition data alone should be cautious in relating ignitions to large fires and subsequent forest composition change in the Pinelands.

Implications of the ecological wildland urban interface

The effects of altered land on wildfire frequency and subsequent forest composition can vary with the area of study. In Barnegat Bay, altered land seems to have a large influence on the frequency of large fires and forest composition change. The area of low fire frequency and the ecological wildland urban interface reaches up to 480m into natural areas where fire frequency directly results in forest composition change (see Table 5). While the initial choice of the buffer distance category was arbitrary (i.e. a 240 m vs. 250 m interval), there does appear to be a measurable gradient in response to distance in that areas closer to altered land have less fire and UPCON to UPDEC transitions. Additionally, because the Mullica River's EWUI is different than Barnegat Bay's, we are confident that our EWUI scales encompass much of the important landscape level variability in this ecosystem. This EWUI (when including both upland and wetland forest) covers 32% of the entire Barnegat Bay watershed (0-480m) and 19% of the Mullica River watershed (0-240m). The effects of altered land on fire frequency in the Mullica River is not as pronounced and may be on a time lag as compared to Barnegat Bay, but Barnegat Bay's altered land density and configuration could serve as a model for potential future ecological affects of altered land on adjacent fire regimes and forest composition in the Mullica River basin. Indeed, we saw that overall, deciduous cover is continuing to increase and coniferous cover is continuing to decrease in the Mullica River watershed for 2007 (Figure 5). Altering water levels in nearby wetlands due to development may also lead to more upland forest within the EWUI (Lathrop et al. 2010). The EWUI is therefore dynamic through time and space. When weather and water levels are considered, it is feasible that the EWUI could expand or contract over time and disturbance dynamics have the potential to increase or decrease in frequency (e.g. in the case of the 1963 drought, the EWUI most likely shrank as fire danger increased).

Wildfire frequencies across the landscape are modified by seasonality, wetland configurations, and proximity to altered land. The influence of fire frequency on forest composition change trajectories provides useful insights into the relative importance of fire frequency in different areas of the Pinelands rather than purely global analyses. Decreases in large fires due to fire suppression throughout New Jersey's fire history have shown that the Pinelands are experiencing fewer large fires overall compared to the early

20th century, and that large fires are becoming exceptionally rare near human altered land in the EWUI. The similarities in fire frequencies between specific forest transitions and areas at specific distances from altered land are striking and point to a likely succession trajectory and pattern for the Pinelands. The EWUI exists due to the historic policy of complete suppression of fires near altered land creating conditions for succession in the Pinelands, so that even though humans cause more ignitions here, large fire frequencies are low. Although suppression is necessary to preserve life and property, it is clear that this policy will result in the gradual loss of the current Pinelands ecosystem. Finding a balance between life and property and ecosystem services will be the main challenge for managers in the future. Currently, new non-fire and pre-fire methods are being considered and tested, such as mechanical thinning (Mike Drake, NJFFS, personal communication).

This study can inform fire and land managers in the Pinelands in several ways:

1. Given the importance of accurate fire records for this ecosystem, this type of information is essential to the NJFFS mission. Records should include accurate prescribed fire perimeters and attributes as well as information on the severity of wildfires. Fire severity can vary within a fire perimeter and can determine future forest composition as well as severity thresholds of fire adapted endangered species.
2. Areas of high upland coniferous to upland deciduous transitions and areas demonstrating the fire frequencies leading to this type of forest transition (e.g. the EWUI area) can be targeted for ecological monitoring and potential loss of habitat for endemic Pinelands species.

3. Areas of upland mixed forest to upland coniferous forest can be studied for the purposes of mimicking the fire regimes leading to this type of transition and halting the continued loss of coniferous forest to deciduous cover, especially in forest preservation areas.
4. Areas where fire has decreased dramatically in more recent years due to proximity to newly altered land can be targeted for prescribed fire and/or fuel reductions in order to address dangerous fuel loads or potential forest composition change.
5. Plans for newly altered land should include an estimate of the effect of the development on future forest fire regimes and composition in adjacent forests by incorporating the spatial extent and severity of the EWUI effect.

The lessons learned in this case study of complex disturbance dynamics in the Pinelands should assist fire managers in designing a comprehensive landscape level fire management plan. Understanding the extreme decrease in hectares burned per year since 1963 and the potential affects on forest composition is also relevant for ecology based management decisions. As less fire occurs in the Pinelands and more areas convert from coniferous to deciduous forest, the potential for the upland forest areas not included in prescribed fire management (27,893 ha of the watershed) in the Barnegat Bay to shift to a deciduous dominated forest is a possibility in less than 200 years (current rate = 143 ha a year across the entire watershed). Presently, fire frequencies and transitions from UPMIX to UPCON would prevent such a change in the middle and interior areas of the Mullica River watershed, but not in EWUI areas. A deciduous or mixed forest would support a different suite of species, and potentially no longer support the species and functions for which the area was originally set aside for as a National Reserve. In addition, Pinelands planners should benefit from the detailed view of fire history in choosing if and where to develop in those areas for which development is currently permitted. Planners should note that any newly altered land will affect the forest composition up to 480m into surrounding forests due to the future probable suppression of fire in these areas. Depending on the configuration of newly altered land, current interior natural areas could be affected as well.

The interactions between human caused LULC change and the ecological consequences that may arise from these influences is apparent in the extent of the ecological WUI defined by the gradient of human influence extending from altered land to interior natural areas. The ecological effects of surrounding development on natural areas are ever-increasing (Kauffman 2004; Leu et al. 2008). Regional estimates of ecological footprints (Leu et al. 2008), national maps of WUI areas (Radeloff et al. 2005) and studies on adjacent disturbance effects (Syphard et al. 2007) show that the areas influenced by humans is extensive, and including the extent to which development influences ecosystem function via the EWUI framework provides an essential decision support system for social and environmental policy. This facilitates a greater understanding of the linkages between the forces of human-caused change and the ecosystem dynamics associated with this change by indicating how the effects of the EWUI further expand the human footprint. Other types of human influenced disturbance, such as intensive use for recreation, biomass harvesting and wild-craft harvesting could also be included in EWUI analyses. The EWUI framework will not only assist local managers in New Jersey but is applicable to other areas of the US and could easily be adapted to any disturbance prone area with large amounts of wildland/urban interface.

Acknowledgements

Funding was provided by the NOAA- Graduate Research Fellowship and the Jacques Cousteau Estuarine Research Reserve. Additionally, funding was provided by the Grant F. Walton Center for Remote Sensing and Spatial Analysis at Rutgers University along with the Graduate Program in Ecology and Evolution, the School for Environmental and Biological Sciences and the Graduate School of New Brunswick. I would like to thank Fred Giliberti and Asa Wright for helping digitize 80 years of fires and John Bognar, Kathy Peirano, and Jim Trimble of the Center for Remote Sensing and Spatial Analysis for their unwavering support. Special thanks to Andrew Windisch for valuable comments on fire history and pine plains data and to Mary Cadenasso for valuable manuscript comments. Thanks also to NJFFS and the USFS Pinelands personnel for providing access to fire history records, local knowledge, and general good advice.

Table 1. Spatial LULC data obtained from the New Jersey Department of Environmental Protections (NJ DEP) for 1986 and 2002 was used to create more general forest and altered land classes. Categories that were subsumed in 2002 categories are indicated (1986 only) as are categories that were created in 2002 (2002 only) from existing 1986 categories.

Forest Categories

Upland Deciduous (UPDEC)	Upland Coniferous (UPCON)
Deciduous forest (>75% crown closure) -1986 Only	Coniferous forest (>75% crown closure) -1986 Only
Deciduous forest (10-50% crown closure)	Coniferous forest (10-50% crown closure)
Deciduous forest (>50% crown closure)	Coniferous forest (>50% crown closure)
Mixed forest (>50% Deciduous with 10-50% crown closure)	Mixed forest (>50% Coniferous with 10-50% crown closure)
Mixed forest (>50% Deciduous with >50% crown closure)	Mixed forest (>50% Coniferous with >50% crown closure)
Deciduous brush/shrubland	Coniferous brush/shrubland
Deciduous/Coniferous forest (>50% Deciduous) - 1986 Only	Coniferous/Deciduous forest (>50% Coniferous) -1986 Only

Upland Mixed (UPMIX)	Atlantic White Cedar (ATWHCED)
Mixed Deciduous/coniferous brush/shrubland	Atlantic White Cedar

Wetland Deciduous (WETDEC)	Wetland Coniferous (WETCON)
Deciduous wooded wetlands	Coniferous wooded wetlands
Deciduous scrub/shrub wetlands	Coniferous scrub/shrub wetlands
Mixed scrub/shrub wetlands (Deciduous dom.)	Mixed scrub/shrub wetlands (Coniferous dom.)
Mixed forested wetlands (Deciduous dom.)	Mixed forested wetlands (Coniferous dom.)

Altered Land Categories

Anderson Level 1	Anderson Level 3 under Wetlands
AGRICULTURE	Wetlands Rights of Way
BARREN LAND	Managed wetland in maintained lawn greenspace
URBAN	Managed wetland in built-up maintained rec area
	Agricultural wetlands (modified)
	Disturbed wetlands (modified)
	Managed wetlands
	Cemetary on wetland -2002 Only

Table 2. Percent forest composition change matrix from 1986 (rows) to 2002 (columns) along with observed minus expected values in terms of gain and loss for the buffer transition area (BTA) closest to altered land (i.e. 0-120m for both dates and an example of 1 of 49 BTA analyses) in the Barnegat Bay watershed. Expected values are computed via Equation 1 for loss from 1986 and a similar equation for gain in 2002 (see methods). Diagonals show persistence of the forest category and off diagonals show shifting forest composition. The UPDEC column in the observed less expected table (in terms of gain) shows the percentage of land gained in 2002 from each forest category. The UPCON row in the observed less expected table (in terms of loss) shows the percentage of land in 1986 lost to each forest category. Because both of these percentages are positive and the highest ranked forest categories in terms of both loss and gain to each other (grey highlight), we conclude that the UPCON to UPDEC transition in this BTA is systematic.

2002

Forest Composition Change Matrix (%)

		UPDEC	UPCON	UPMIX	WETDEC	WETCON	ATWHCED	Total 1986	Loss from 1986
1986	UPDEC	13.97	1.30	0.34	0.89	0.17	0.02	16.70	2.73
	UPCON	6.02	36.77	0.67	0.65	0.76	0.20	45.08	8.30
	UPMIX	0.54	0.67	0.76	0.06	0.02	0.00	2.06	1.29
	WETDEC	1.00	0.59	0.07	16.38	2.60	0.48	21.13	4.74
	WETCON	0.25	1.07	0.03	1.94	8.49	0.26	12.04	3.55
	ATWHCED	0.04	0.14	0.01	0.40	0.13	2.29	3.01	0.72
	Total 2002	21.83	40.54	1.88	20.32	12.17	3.26	100.00	21.33
	Gain to 2002	7.86	3.77	1.12	3.94	3.68	0.97	21.33	

Expected (in terms of gain in 2002)

		UPDEC	UPCON	UPMIX	WETDEC	WETCON	ATWHCED
1986	UPDEC	13.97	1.15	0.19	0.83	0.70	0.17
	UPCON	4.25	36.77	0.51	2.25	1.89	0.45
	UPMIX	0.19	0.14	0.76	0.10	0.09	0.02
	WETDEC	1.99	1.45	0.24	16.38	0.88	0.21
	WETCON	1.14	0.83	0.14	0.60	8.49	0.12
	ATWHCED	0.28	0.21	0.03	0.15	0.13	2.29

Observed - Expected (in terms of gain in 2002)

		UPDEC	UPCON	UPMIX	WETDEC	WETCON	ATWHCED
1986	UPDEC	0.00	0.16	0.15	0.06	-0.53	-0.14
	UPCON	1.77	0.00	0.15	-1.60	-1.13	-0.25
	UPMIX	0.35	0.53	0.00	-0.04	-0.07	-0.02
	WETDEC	-0.99	-0.86	-0.17	0.00	1.72	0.27
	WETCON	-0.88	0.25	-0.11	1.34	0.00	0.14
	ATWHCED	-0.24	-0.07	-0.02	0.25	0.01	0.00

Expected (in terms of loss from 1986)

		UPDEC	UPCON	UPMIX	WETDEC	WETCON	ATWHCED
1986	UPDEC	13.97	1.41	0.07	0.71	0.42	0.11
	UPCON	3.05	36.77	0.26	2.84	1.70	0.45
	UPMIX	0.29	0.53	0.76	0.27	0.16	0.04
	WETDEC	1.30	2.41	0.11	16.38	0.72	0.19
	WETCON	0.88	1.64	0.08	0.82	8.49	0.13
	ATWHCED	0.16	0.30	0.01	0.15	0.09	2.29

Observed - Expected (in terms of loss from 1986)

		UPDEC	UPCON	UPMIX	WETDEC	WETCON	ATWHCED
1986	UPDEC	0.00	-0.11	0.27	0.18	-0.26	-0.09
	UPCON	2.98	0.00	0.41	-2.19	-0.94	-0.26
	UPMIX	0.26	0.13	0.00	-0.21	-0.14	-0.04
	WETDEC	-0.30	-1.82	-0.04	0.00	1.88	0.29
	WETCON	-0.63	-0.57	-0.04	1.12	0.00	0.13
	ATWHCED	-0.12	-0.16	0.00	0.25	0.04	0.00

Table 3. Matrix of buffers widths (in meters) in 1986 (rows) and 2002 (columns) and hectares associated with all combinations of distance from altered land into forested areas for Barnegat Bay (72,133 ha) and Mullica River (104,697 ha) watersheds. Diagonals represent areas of forest retaining the same proximity to altered land for both years. Areas above the diagonal were closer in proximity to altered land in 1986 and transitioned to a category further away from altered land in 2002 due mainly to a former unspecified brushland succeeding into an identifiable forest type and 'filling in' a previously non-forested area. Areas below the diagonal were farther away from altered land in 1986, but due to newly altered land in the intervening years the proximity of these forested areas was closer to altered land in 2002 (transitioning to a category closer to altered). Buffer transition areas in grey were the BTAs for which fire frequency was enumerated (see category definitions in methods). Areas delineated with the bold black square (0-480m for Barnegat Bay and 0-240m in Mullica) are the forested areas within the ecological wildland-urban interface.

		2002						
		Barnegat Watershed						
		0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
1986	0-120	19880.73	2004.57	834.39	545.40	199.98	32.85	0.00
	120-240	3250.17	6423.48	1707.21	709.74	233.82	43.20	0.00
	240-480	1380.69	1675.89	9070.20	2612.88	540.00	84.51	0.00
	480-1080	416.70	460.26	1581.39	11261.88	1523.07	215.55	22.23
	1080-1680	7.74	11.97	50.49	565.65	2706.48	406.44	106.56
	1680-2280	0.00	0.00	0.00	0.00	69.39	932.76	169.38
	2280+	0.00	0.00	0.00	0.00	0.63	10.89	383.67
		Mullica watershed						
		0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
1986	0-120	17601.21	1859.22	526.05	261.18	61.29	61.2	0.72
	120-240	2156.13	8144.1	1710.45	405.81	109.35	81.9	7.56
	240-480	882.72	1587.51	14012.19	2224.53	416.07	176.49	48.33
	480-1080	505.62	548.19	1950.93	21515.13	2272.5	475.38	257.58
	1080-1680	165.87	145.71	405.9	1833.57	9777.42	857.25	432.9
	1680-2280	88.29	58.41	147.78	524.25	849.78	4030.65	771.93
	2280+	44.82	11.34	11.97	138.24	444.96	633.6	3463.11

Table 4. Example of gain vs. loss for percent forest composition change in one buffer transition area (closest to altered land for both 1986 and 2002; 0-120m) in the Barnegat Bay watershed. The percent difference is used to calculate total hectares gained (positive) and lost (negative) in each forest category by multiplying it by the total BTA area. These areal changes are displayed in Table 5 for UPDEC, UPCON and UPMIX in all BTAs with systematic changes.

	Gain	Loss	Difference
UPDEC	7.86	2.73	5.13
UPCON	3.77	8.30	-4.53
UPMIX	1.12	1.29	-0.17
WETDEC	3.94	4.74	-0.80
WETCON	3.68	3.55	0.13
ATWHCED	0.97	0.72	0.25
Total	21.33	21.33	0.00

Table 5. Hectares of gain and loss for upland forest categories with systematic changes greater than 10 hectares are shown for each buffer transition area (deciduous = UPDEC, coniferous = UPCON, mixed = UPMIX). Rows represent proximity to altered land in 1986 and columns represent proximity to altered land in 2002. EWUIs (where deciduous gains come directly from coniferous and/or mixed losses due to fire frequency changes) are boxed for Barnegat Bay (0-480m) and Mullica River (0-240m).

2002

Barnegat watershed

Deciduous Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	1020.42	88.92	25.38				
120-240	187.56	210.60	47.97				
240-480	89.19	37.38	202.32	21.69			
480-1080	28.53	15.66	60.93	234.18			
1080-1680				12.15			
1680-2280							
2280+							

Coniferous Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	-901.44	-53.73					
120-240	-200.79	-154.71		17.82			
240-480	-96.75		-48.51	104.85	32.04		
480-1080	-18		-53.1	200.52	59.67		
1080-1680					162.27		
1680-2280						-20.43	
2280+							

Mixed Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	-34.56						
120-240			-10.08				
240-480	26.46		-42.57	-50.76			
480-1080			14.67	-207.63	-29.34		
1080-1680					-115.38		
1680-2280						37.26	
2280+							

2002

Mullica watershed

Deciduous Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	350.26	33.47					
120-240	36.65	100.99	9.24		26.03		
240-480	11.48	26.99	53.25	30.92			
480-1080	12.42		22.83	40.88	28.86	-17.10	-37.86
1080-1680	22.23		17.82	30.03		-36.54	-14.40
1680-2280				35.91	24.12	-11.34	-10.71
2280+							-40.95

Coniferous Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	-40.4828						
120-240	-20.052		18.81495				
240-480			123.3073	28.25153			
480-1080				398.0299		21.05933	49.19778
1080-1680	-10.08			-20.7	274.95	67.68	33.03
1680-2280				-17.1	17.37	220.59	42.39
2280+					14.4	18.18	171.18

Mixed Change

1986	0-120	120-240	240-480	480-1080	1080-1680	1680-2280	2280+
0-120	-207.694	-15.75					
120-240	-12.0743	-71.6681	-9.57852	-10.5105			
240-480		-15.383	-98.0853	-22.6902			
480-1080	19.163		15.19774	-87.5666	-27.27		
1080-1680	14.31	8.19		21.69	-136.62	-27.09	
1680-2280					-36.63	-107.19	-19.08
2280+						-15.48	-51.84

Figure captions

Figure 1. Study area depicting the New Jersey Pinelands National Reserve along with the Barnegat Bay and Mullica River watersheds. Additionally, all digitized fire perimeters in the fire geodatabase are shown with prescribed fire in blue and wildfire in red. Perimeters are not a complete dataset of paper wildfire records for New Jersey and the dataset was focused on counties falling within our watersheds. Only wildfires overlapping the Barnegat Bay and Mullica River watersheds were analyzed. Divisions A, B and C are shown for reference to wildland fire fighting jurisdictions. No spatial prescribed fire records were available for Division C, although prescribed fire is also known to be conducted in this area

Figure 2. Buffers of altered land showing distance to altered land within a forested area for a) 1986 and b) 2002. The last panel c) depicts the intersection of these two dates to form buffer transition areas (BTAs) which reference both distance to altered land in 1986 and distance to altered land in 2002. Areas such as old fields were not buffered as altered land, but were also not included in the forest composition change enumeration. As in c) the white area not included in the BTA scene is larger than the white area representing altered land in b) due to the presence of old fields or other non-forest/non-altered classes in the scene for either date. In c), the arrow points to a buffer transition area **transitioning toward altered land**.

Figure 3. Wildfire frequency map of the study area from 1924-2002 showing the area of the dwarf pine plains along with a location map of New Jersey, USA and the Mullica River and Barnegat Bay watersheds of the New Jersey Pinelands

Figure 4. Hectares of large wildfires by year (1924-2007) in Barnegat Bay and Mullica River watersheds. Years with no large fires are more frequent after 1963, (although 1975-1978 data are lacking).

Figure 5. Total change in upland coniferous, deciduous and mixed forest change for Barnegat Bay (a) and Mullica River (b) watersheds from 1986-2007. Although percent of total forest shows a decline in coniferous forest cover, overall changes mask the importance of the location of the change and an understanding of how composition may change along the gradient from human altered land to interior natural areas is lost.

Figure 6. A comparison of large fire frequencies per a) forest composition categories (total area per category = 100%) and b) buffer transition areas in the Barnegat Bay watershed (1986-2002).

Figure 7. A comparison of large fire frequencies per a) forest composition areas (total area per category = 100%) and b) buffer transition areas in the Mullica River watershed (1986-2002).

Figure 8. Ignitions per hectare for buffer transition areas within a) Barnegat Bay and b) Mullica River watersheds. Although many points may ignite, few fires gain large fire status (>40.47 ha or 100 ac; 1986-2002).

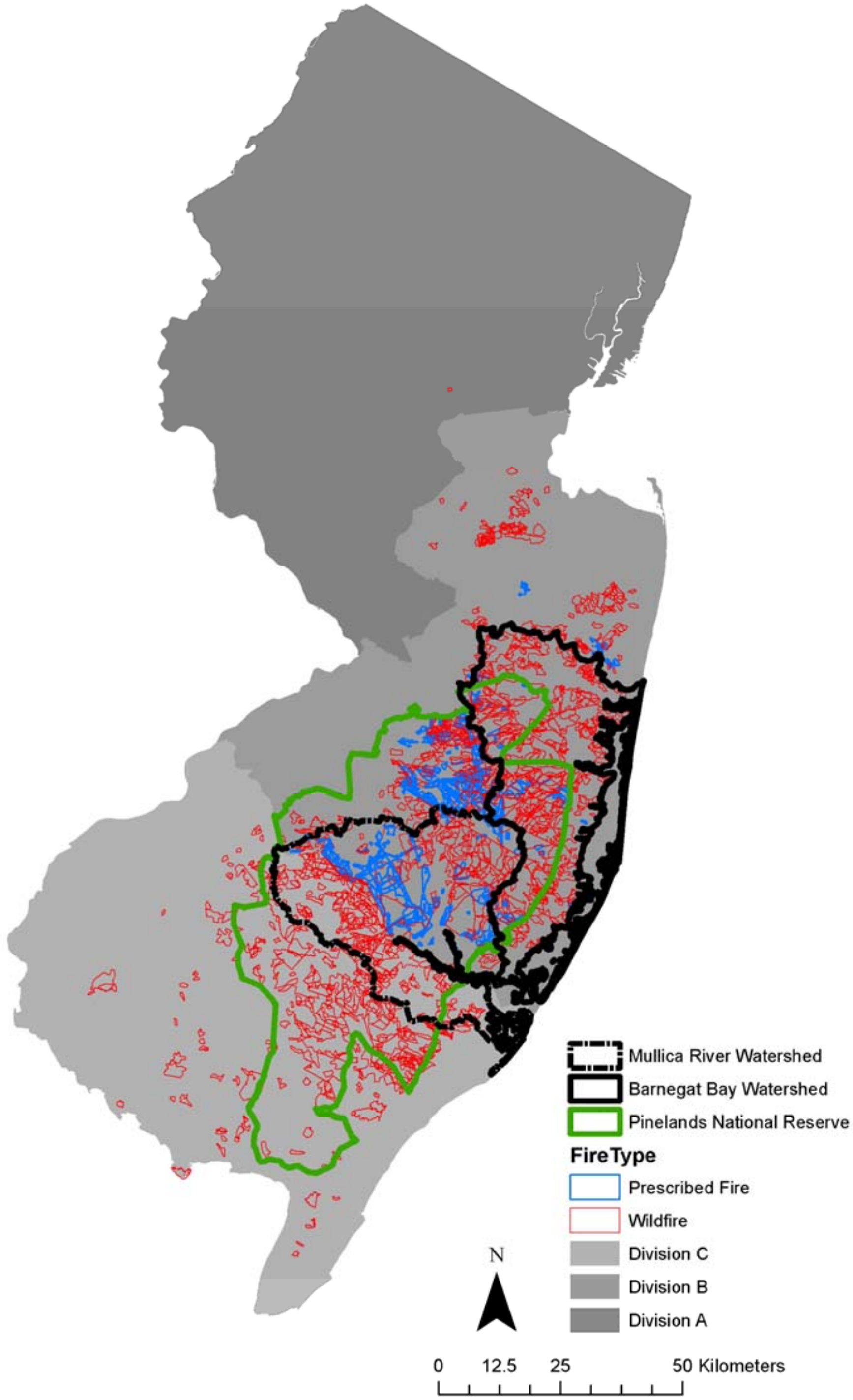

Mullica River Watershed
Barnegat Bay Watershed
Pinelands National Reserve
FireType
Prescribed Fire
Wildfire
Division C
Division B
Division A
N
0
12.5
25
50 Kilometers

Figure 1.

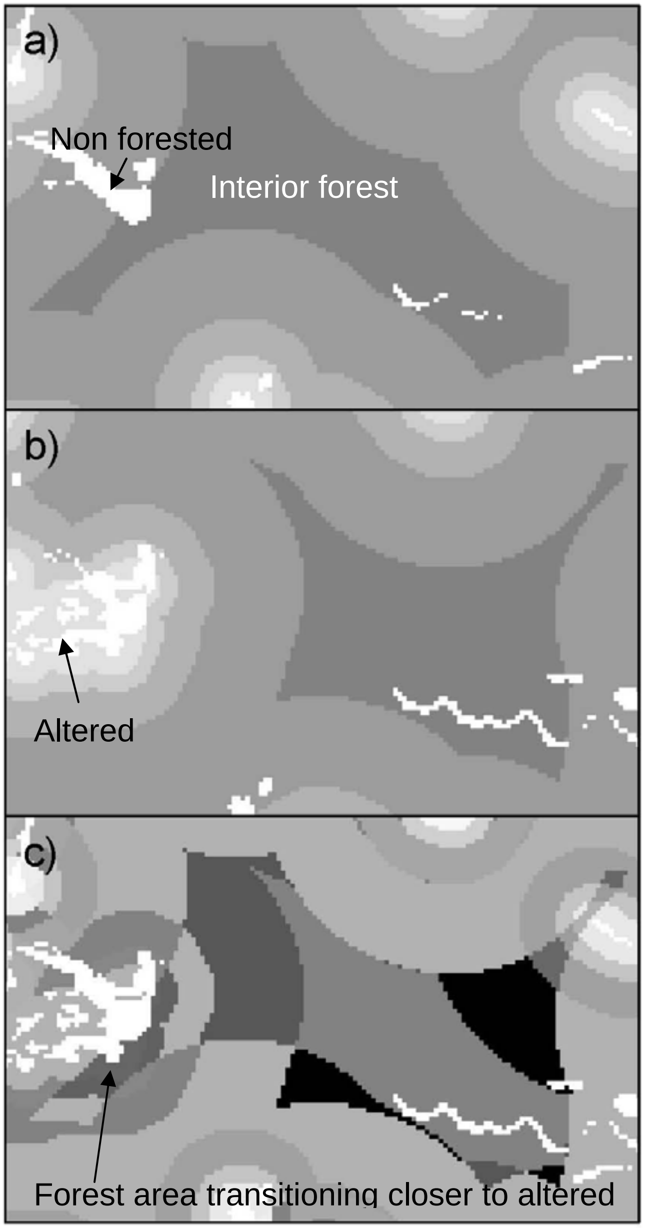

a)
Non forested
Interior forest
b)
Altered
c)
Forest area transitioning closer to altered

Figure 2.

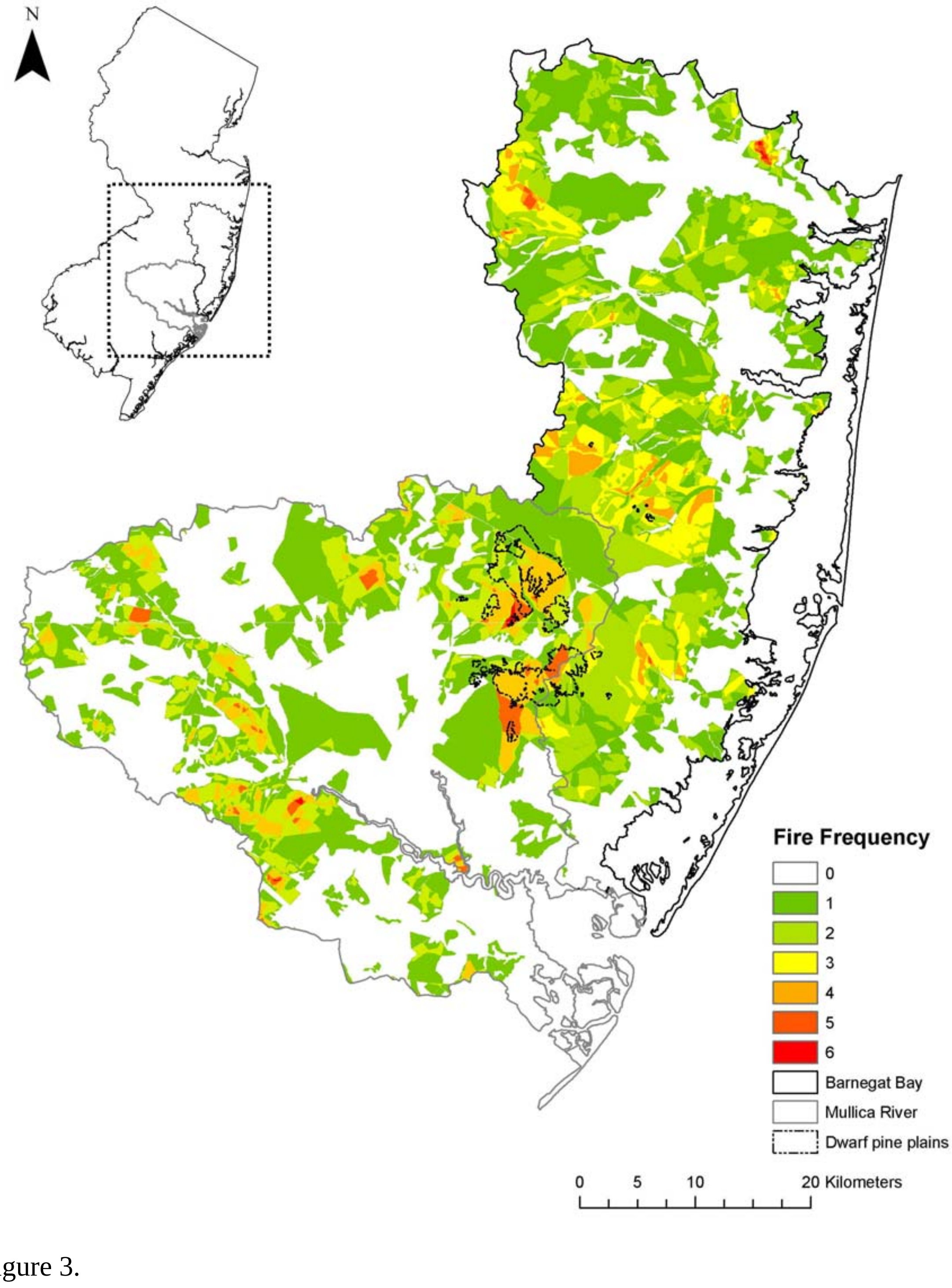
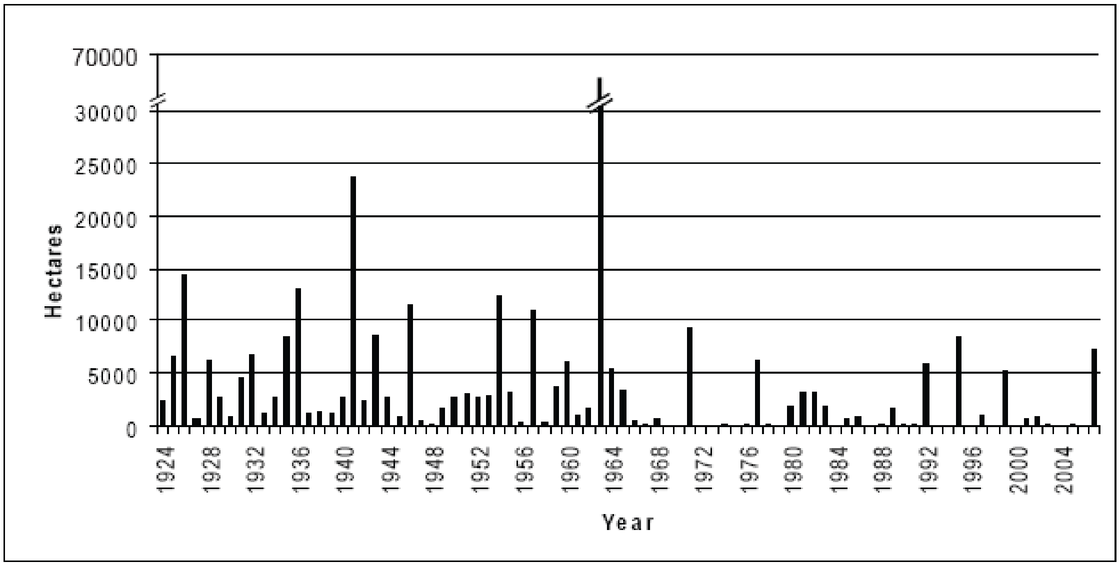

N
Fire Frequency
0
1
2
3
4
5
6
Barnegat Bay
Mullica River
Dwarf pine plains
0
5
10
20 Kilometers

Figure 3.



70000
30000
25000
20000
15000
10000
5000
0
Hectares
1924
1928
1932
1936
1940
1944
1948
1952
1956
1960
1964
1968
1972
1976
1980
1984
1988
1992
1996
2000
2004
Year

Figure 4.

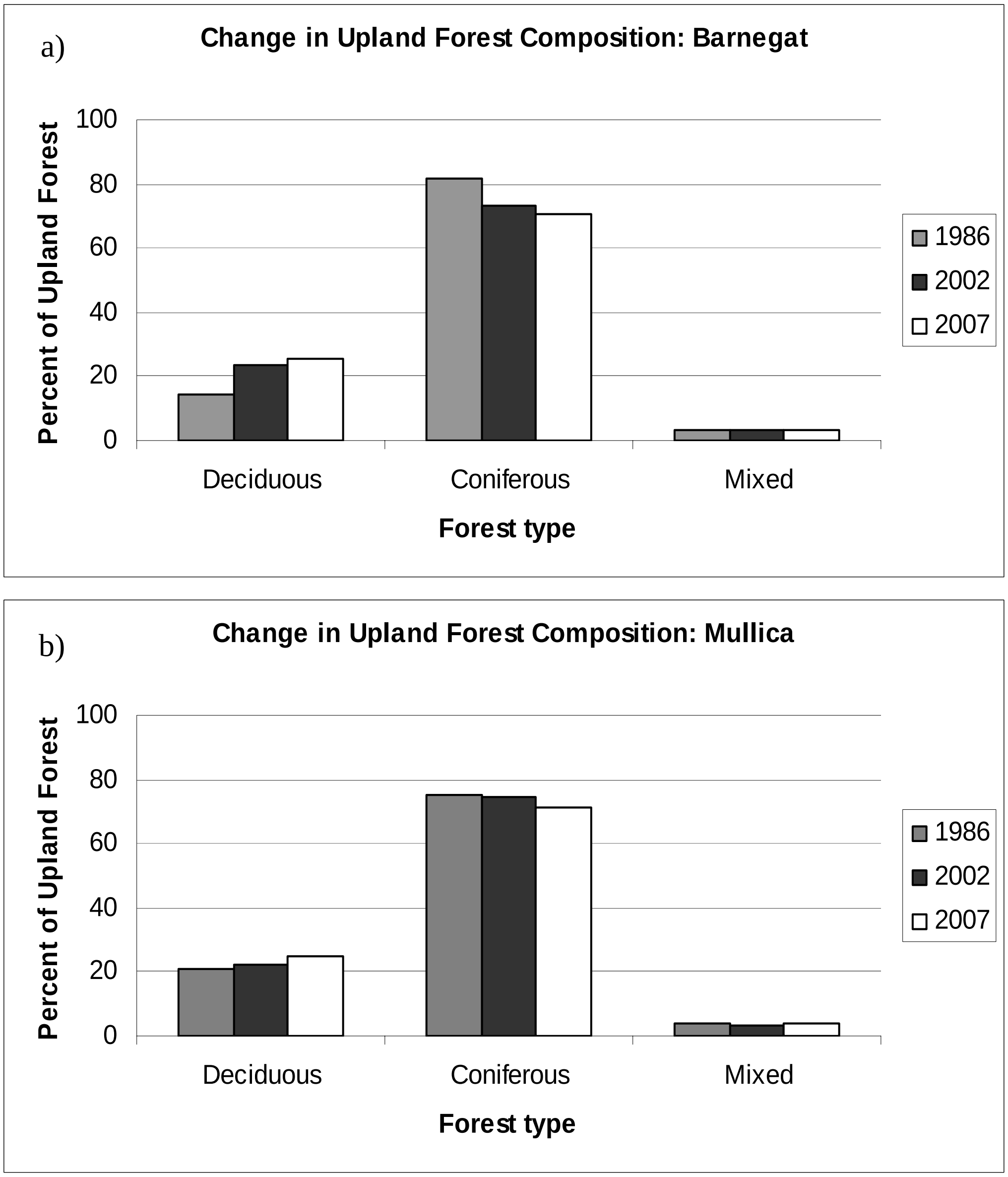

a)
Change in Upland Forest Composition: Barnegat
Percent of Upland Forest
100
80
60
40
20
0
1986
2002
2007
Deciduous
Coniferous
Mixed
Forest type
b)
Change in Upland Forest Composition: Mullica
Percent of Upland Forest
100
80
60
40
20
0
1986
2002
2007
Deciduous
Coniferous
Mixed
Forest type

Figure 5.

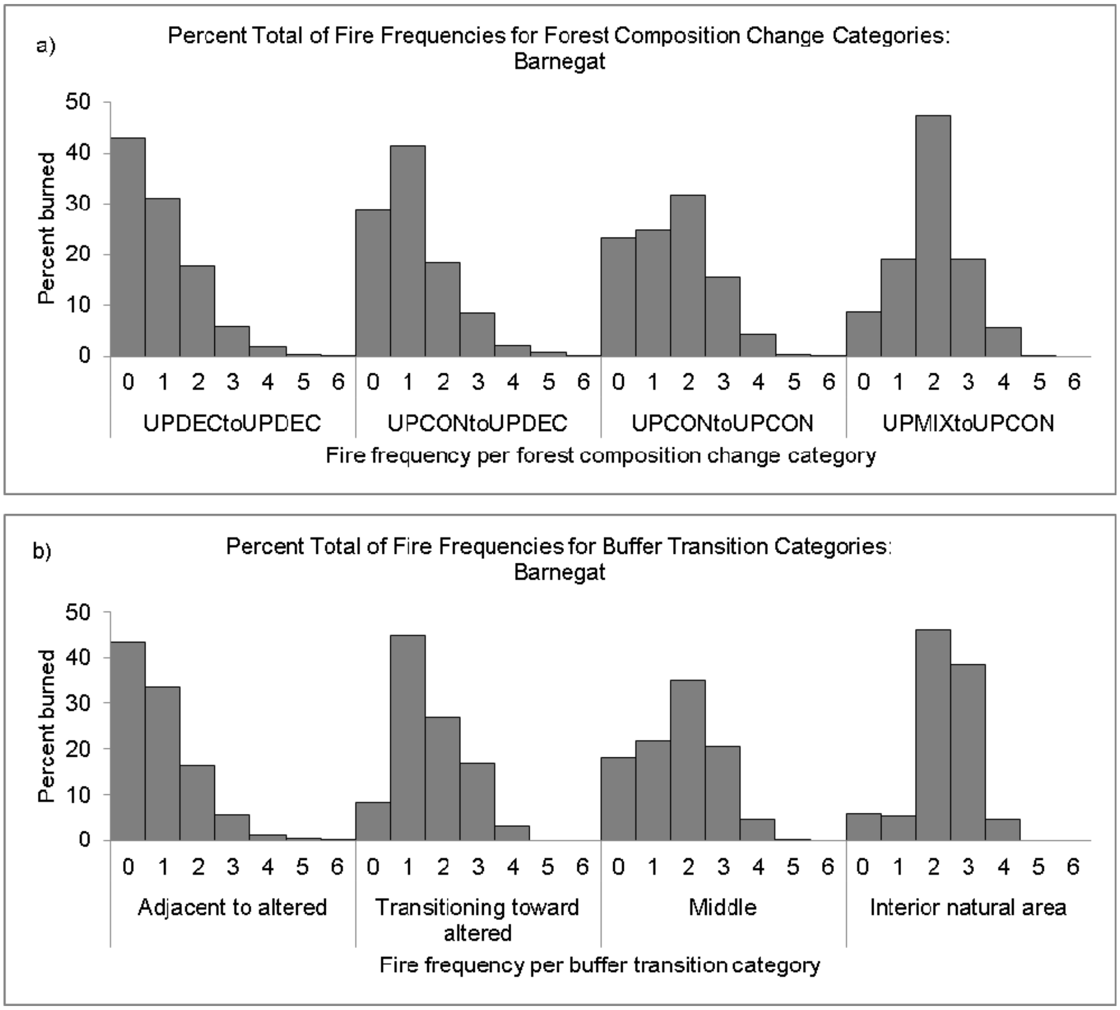
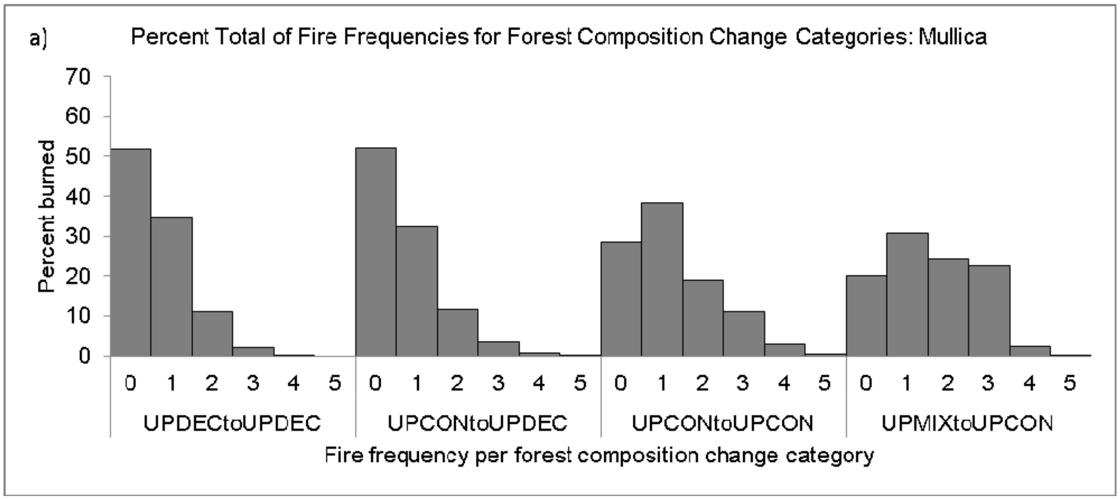

a)
Percent Total of Fire Frequencies for Forest Composition Change Categories:
Barnegat
Percent burned
50
40
30
20
10
0
0 1 2 3 4 5 6 0 1 2 3 4 5 6 0 1 2 3 4 5 6 0 1 2 3 4 5 6
UPDECtoUPDEC
UPCONtoUPDEC
UPCONtoUPCON
UPMIXtoUPCON
Fire frequency per forest composition change category
b)
Percent Total of Fire Frequencies for Buffer Transition Categories:
Barnegat
Percent burned
50
40
30
20
10
0
0 1 2 3 4 5 6 0 1 2 3 4 5 6 0 1 2 3 4 5 6 0 1 2 3 4 5 6
Adjacent to altered
Transitioning toward altered
Middle
Interior natural area
Fire frequency per buffer transition category

Figure 6.


a)
Percent Total of Fire Frequencies for Forest Composition Change Categories: Mullica
Percent burned
70
60
50
40
30
20
10
0
0 1 2 3 4 5 0 1 2 3 4 5 0 1 2 3 4 5 0 1 2 3 4 5
UPDECtoUPDEC
UPCONtoUPDEC
UPCONtoUPCON
UPMIXtoUPCON
Fire frequency per forest composition change category

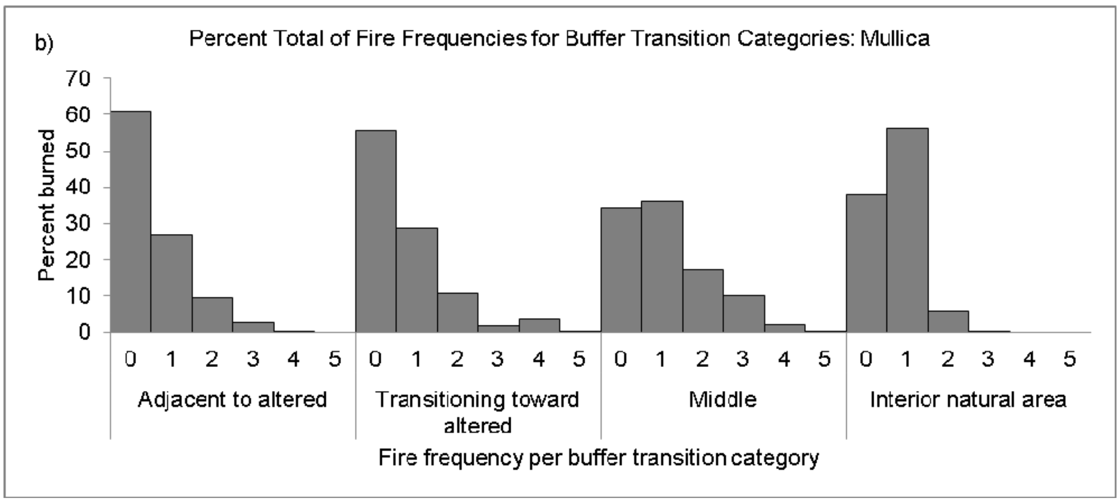
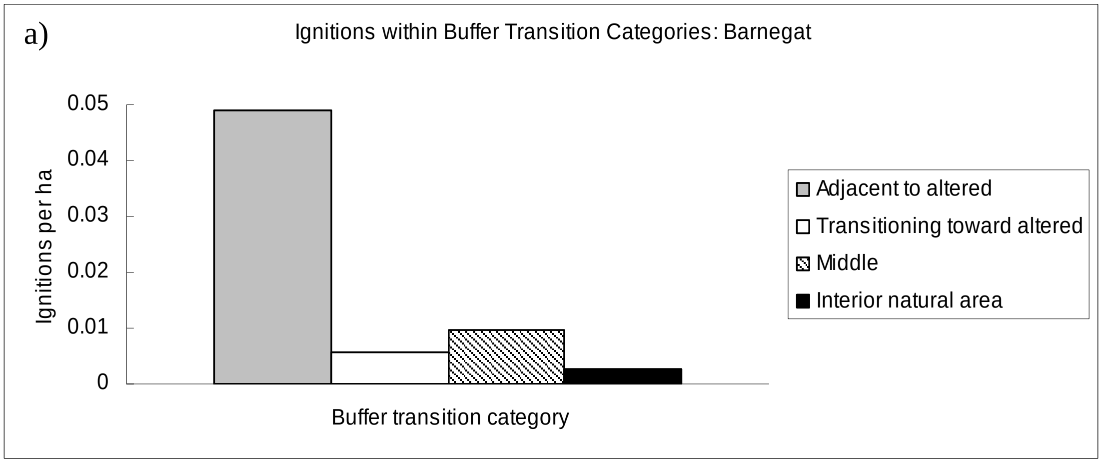

b)
Percent Total of Fire Frequencies for Buffer Transition Categories: Mullica
Percent burned
70
60
50
40
30
20
10
0
0 1 2 3 4 5 0 1 2 3 4 5 0 1 2 3 4 5 0 1 2 3 4 5
Adjacent to altered
Transitioning toward altered
Middle
Interior natural area
Fire frequency per buffer transition category

Figure 7.


a)
Ignitions within Buffer Transition Categories: Barnegat
0.05
0.04
0.03
0.02
0.01
0
Ignitions per ha
Buffer transition category
Adjacent to altered
Transitioning toward altered
Middle
Interior natural area

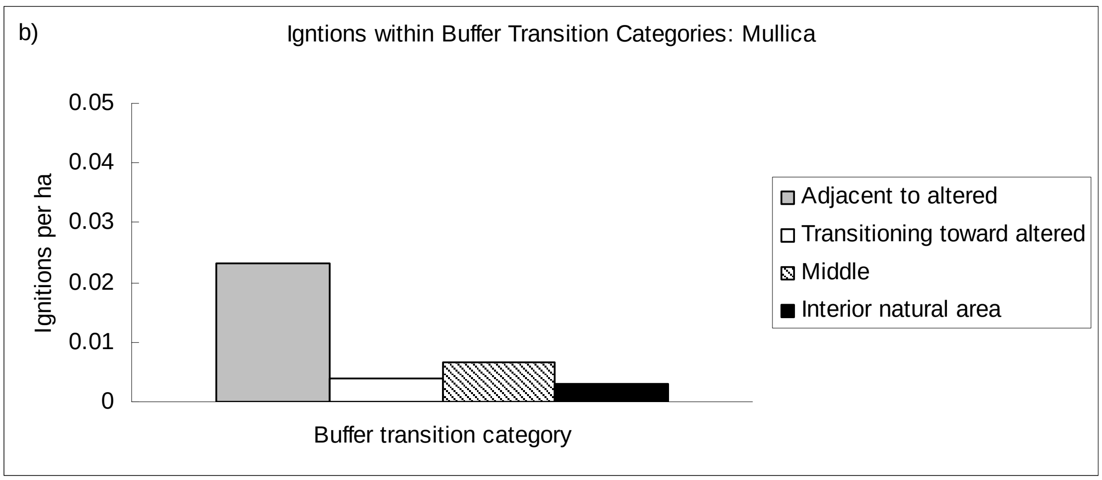

b)
Igntions within Buffer Transition Categories: Mullica
0.05
0.04
0.03
0.02
0.01
0
Ignitions per ha
Buffer transition category
Adjacent to altered
Transitioning toward altered
Middle
Interior natural area

Figure 8.

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CHAPTER 2

Investigating water quality as a result of large fires in the Jacques Cousteau National Estuarine Research Reserve watersheds

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Abstract:

This is the first study to address the effects of wildfire on water quality downstream of fire events in the Pinelands of New Jersey. The Barnegat Bay and Mullica River watersheds lead into the Jacques Cousteau National Estuarine Research Reserve and provide much of the clean water for southern and coastal New Jersey. Nutrient availability is a key focus of eutrophication concerns in the Barnegat watershed. To address this question, we employed data from an existing network of water quality/quantity monitoring stations that collect a suite of water quality parameters on a periodic basis. It should be noted that the monitoring station location and the timing of sample acquisition were not explicitly designed to study wildfire influences on water quality. While data on nutrients such as nitrogen were of special interest in our study, these were not collected on a wide enough scale to be useful; rather, only water quality parameters such as pH, specific conductance and turbidity were available on a regular basis. Using a geospatial database of forest fire events, we constructed several tests of fire effects on water quality measures including pH, specific conductance and turbidity which can indicate nutrient availability. We investigated whether parameters were significantly different before and after a fire and compared fire year differences to non-fire year differences. Additionally, we addressed whether spatial variables, such as network stream distance to fire and the percent of each drainage basin burned were important aspects of the magnitude of change in each parameter. We found only a small number of instances where there was adequate pre- and post-fire water quality data leading to a comparatively small sample size. Likewise, the spatial extent of a wildfire within a gauged basin was also small (i.e., the highest % of the basin burned was less

than 10%). Overall, we found no evidence that fire leads to significant changes in pH, specific conductance and turbidity or that our spatial indices are influential in determining parameter values. However, we can not conclude definitively whether wildfire does or does not have a substantive impact on Pinelands water quality as our study was hampered by the small sample size of fires that had water quality data before and after the fire, as well as the lack of major fire events on a watershed % basis. Dilution from the influx of groundwater base flow and plant uptake of nutrients during the growing season may also be contributing factors to insignificant changes for water quality measurements in response to fire. In comparison to long term land-use land-cover change, the immediate effects of fire on water quality were observed to be minor. Further research should involve intensive water testing down stream after fire to improve our ability to analyze the effects of fire on water quality measures, although comparisons to non-fire years requires continued testing on a regular basis.

Introduction

Landscape level forest wildfires have been shown in some areas to affect water quality over the short and long term and at different spatial scales (Bladon et al. 2008; Carignan et al. 2000; Minshall et al. 2001). Wildfire effects on water quality may or may not exacerbate land-use/ land-cover (LULC) effects within large watersheds. The relative amount of change to water quality that wildfire contributes in comparison to LULC change will indicate the ecological implications of different fire management trajectories and the potential additive affects of different types of landscape change. If wildfire alone is not a large contributor to water quality changes through time and space, then fire managers may have more flexibility in their approach to the creation of prescribed fire management plans.

The New Jersey Pinelands, a unique fire dependent ecosystem in a highly urbanized state, contain the watersheds upstream of the Jacques Cousteau National Estuarine Research Reserve (JC NERR). This area was designated for long term research, monitoring and education and is an example of one of the least disturbed estuaries in the mid-Atlantic region (Kennish 2007). The two major watersheds in the pinelands are the Barnegat Bay and Mullica River watersheds which were also protected as part of a National Reserve in 1978 to enhance protection of the Kirkwood - Cohansey aquifer, (comprising approximately 17 trillion gallons (Walker and Solecki 1999)), and to limit development pressures from the adjacent New York City and Philadelphia metropolitan areas. The aquifers in the NJ pinelands are also an important clean water source for Southern New Jersey (Kauffman et al. 2001).

Human alteration of habitat and water quality is a key management issue for the watersheds contributing to the JC NERR (JCNERR 2009), although loss of forest land to development has slowed since the implementation of the regionally based comprehensive Pinelands management plan and has been consistent with the plan itself (Bunnell et al. 2003; Walker and Solecki 1999). However, even with the rate of development slowing, there has been an increase in the total area, as well as the number of patches of developed areas, managed grasslands, and barren land (Hasse and Lathrop 2010). Lathrop and Conway (2001) found that in the Barnegat Bay watershed, regardless of the scenario of regulations used (including the current development plan), water demand will unfortunately exceed supply and water quality will be impacted, primarily due to the high levels of impervious surface. Additionally, with increased groundwater withdrawal due to development pressures associated with well-drilling, GIS modeling has demonstrated that drawdown of the water table will result in the loss of wetlands and wetlands forest habitat (Lathrop et al. 2010).

LULC disturbance in the Pinelands is known to contribute to failing ecosystems downstream (Baker and Hunchak-Kariouk 2007; Zampella et al. 2007a). One study demonstrated disturbance effects on streambank vegetation composition, with non-native flora indicative of disturbed watersheds (Zampella and Laidig 1997). Residential and agricultural development also led to the replacement of distinctively Pinelands flora with non-indigenous species (Morgan and Philipp 1986). These studies were confirmed in a more recent ecological integrity survey which included the effects of landcover change on vegetation, anuran populations, fish assemblages and water quality measures (Zampella et al. 2006).

Anthropogenic landscape level watershed disturbance such as land-use/ land cover change has a rich history of study in relation to downstream water quality measurements and associated ecosystem change. However, disturbance to watersheds can also come in the form of wildfire. In general, the timing and scale of these disturbances can determine their effects on downstream ecology via water quality. Fire is an important disturbance in the Pinelands of New Jersey and historically, the fire regime has decreased from an average of 3985 ha a year in the 1930s to an average of 2614 ha a year in the 1990s. It has also become much more sporadic in area burned each year, with many calendar years experiencing no large fires at all after 1963.

The focus of this research was to investigate long term, watershed scale effects of fire on water quality in the JC NERR watersheds in the Pinelands of New Jersey and to assess the magnitude of the effect of wildfire in comparison to LULC change. The relationships between fire and water quality in the JC NERR watersheds have yet to be addressed in a comprehensive manner and the effect of large fires on water quality within these watersheds is lacking in published literature. Understanding the relationships between fire and water quality at the landscape scale with the best available spatial and temporal data is a first step in creating ecologically viable long term fire management plans for the area, and was the goal of this study.

Our objective was to conduct exploratory data mining to determine whether any fire effect signal in relation to water quality can be found and to evaluate the utility of the existing data set to quantify that signal. We investigated historic fire events in the JC NERR watersheds for comparison with downstream water quality measurements. Our hypothesis was that water quality measures such as pH, specific conductance (SC), and turbidity increase after fire, but that these parameters are also dependent on the size of fire and the distance to the monitoring station. The focus on these parameters as opposed to nutrient parameters such as nitrogen and phosphorous was necessary due to data limitations. However, pH and SC have been shown to be directly related to nutrient levels (see *Fire effects on water quality measurements* below). Results of this study will assist managers in understanding whether we can determine the benefits or susceptibilities of proposed fire management plans to the watersheds as well as the estuaries downstream. For example, will wildfire present an exacerbating effect on declining water quality when combined with known LULC change effects? Also, is there a threshold at which fire will affect the ability of the watershed to provide needed water quality services?

Background

NJ Pinelands water quality background

Water input into the pinelands system is composed of precipitation, with over half of output lost to evapotranspiration. The Cohansey sand formation lying above the aquifer is highly permeable and the remainder of the water budget recharges groundwater stores and overflows into surface streams (Kauffman et al. 2001). The Barnegat Bay watershed is highly developed in the North and impervious surface is expected to increase here (Lathrop and Conway 2001). The Mullica River watershed is one of the most undeveloped watersheds in the densely populated northeast corridor of the US (Kennish 2007).

In general the Pinelands stream system is highly oligotrophic (Wang 1984) and soils are conducive to mineral acid (low pH) stream water conditions from sulfate sources (Morgan 1991). pH tends to trend lower with higher flow due to the flushing of bogs and swamps common in the Pinelands. Morgan (1984) noted that elevated pH is of concern in the Pinelands due to agricultural and residential inputs resulting in changes to aquatic plant and animal communities. Specific conductance generally increases as one moves from freshwater headwaters to the tidal estuaries along Barnegat Bay and Mullica River's Great Egg Harbor but has also been shown to increase in response to increased altered land cover (Zampella et al. 2007b). Turbidity is the least studied stream parameter of the three mentioned here for our area but has been shown to be quite high in

Barnegat Bay mainly due to the eutrophic conditions, restricting the ability for sea grass to thrive (Kennish et al. 2007).

Fire effects on water quality parameters

Fire can contribute to changes in pH, specific conductance and sediment load through several pathways but is mainly tied to changes in soils mediated by fire severity (Ice et al. 2004). Neary et al (2005) present a review of studies relating fire to water quality parameters. They found that most studies showed only a slight increase in pH which can be attributed to ion exchange with H+ from cations leaching into soil from ash deposits (Miller and Findley 2001; National Wildfire Coordinating Group 2001; Neary et al. 2005). An increase in pH leads to an increase in nitrogen and phosphorous cycling which can also have implications for downstream flora and fauna that are normally nitrogen or phosphorous limited. These affects are typically significantly higher in low pH sites (such as the Pinelands, Ministry of Agriculture Food and Fisheries and British Columbia 2004; National Wildfire Coordinating Group 2001; Neary et al. 2005). Leaching of base cations can continue for up to three years after wildfire which raises the pH of infiltrating soil water (Boerner and Forman 1982)

Ash deposits from wildfire also leads to a higher Na^+ content in stream water, increasing specific conductance (SC) which is in turn an indicator of increasing nutrient levels. Flora and fauna accustomed to lower salinity and/or nutrient levels are significantly affected under these conditions (Ministry of Agriculture Food and Fisheries and British Columbia 2004; Neary et al. 2005). Higher nutrient levels (of elements such as nitrogen and phosphorous) could lead to eutrophication (low oxygen levels) via algal blooms downstream or in the estuary.

Reduced to infiltration rates and increased runoff of minerals due to the hydrophobic affects of burned soils is common (Ice et al. 2004) Increases in sediment loads and the higher turbidity associated with increased runoff are common after wildfire on steeper slopes (Helvey 1980). Turbidity on more moderate slopes (as in the Pinelands) may not change unless there is a significant input of ash via aerial input (Ministry of Agriculture Food and Fisheries and British Columbia 2004; National Wildfire Coordinating Group 2001; Neary et al. 2005). Effects of turbidity on stream biota could range from difficulty finding prey to fish gill function (Bash et al. 2001).

Historically, prescribed fires have been more readily studied due to advanced planning advantages in research; therefore a background of these studies is an important part of our understanding of the effects of fire on water quality in general. Richter's (1982) study on the effects of prescribed fire on water quality and nutrient cycling showed limited effects on soil and hydrological systems, including SC and pH in groundwater of coastal pine forests. It should be noted, however, that the author's study of paired watershed regression models used a total of two watersheds (one pair) and incorporated no spatial or temporal autocorrelation considerations. Beche et al (2005) found that prescribed fire increased water chemistry measures including sulfate and total phosphorous with recovery recorded in less than one year using a before/after control/impact test (BACI). This study investigated only one prescribed fire with low to moderate fire severity with a small area burned (26 ha). Another study concerning prescribed fire in the piedmont of South Carolina found that burns did not affect storm runoff, sediment concentration or export between four pairs of control and impact watersheds. In addition, they found no significant changes in cations from prescribed fire. The study used repeated measures ANOVA but did not take spatial considerations into account. A study of Appalachian pine-hardwood forests used felling and prescribed fire to assess watershed effects (Knoepp and Swank 1993). This study sampled three paired watersheds but only one stream was suitable for use for streamwater samples. A small increase in nitrate was detected in the streamwater of the prescribed burn area.

In comparison to prescribed fire, there are relatively few studies concentrating on the relationship between wildfire and water quality measures. Overall, wildfire tends to be more severe (has higher heat levels) than prescribed fire and results in higher levels of mineral runoff due to burning of soil organic matter and aboveground biomass (Brady and Weil 2008). Lane (2008) found that phosphorous and nitrogen increased 5 to 6 fold after wildfire and that the first year after fire was the most relevant in terms of high nutrient levels. Bladon et al (2008) found elevated levels of dissolved organic nitrogen (DON), total nitrogen and nitrate after one wildfire in the southern Rocky mountains of Alberta lasting up to three years post-fire. For lakes, Carignan et al (2000) found up to a sixty fold increase in mobile ions for burned boreal lakes and in most cases these increases were directly related to the total area burned. Changes in water quality from one fire of 26,000 ha were evaluated over five different streams in Idaho (Minshall et al. 2001). This study found an increase in nutrients and fine sediments after fire and over the next season, but cations were stable throughout this time. Small disturbances, such as rainfall, affected burned streams more readily than unburned streams.

In our study areas, Zampella and others (Pinelands Commission 1981, revised 2010; Zampella 1994; Zampella et al. 2006; Zampella et al. 2007a; Zampella et al. 2007b) have conducted a series of studies on the effects of LULC change and altered land disturbance on water quality in the Pinelands. They have consistently found that higher development levels lead to rises in pH and SC with a significant interaction between the two variables. However, fire disturbance has not been taken into account in these studies. Also in our study area, Morgan (1984) surmised that the wildfires of the epic year of 1963, when 74,000 ha burned after a severe drought, were the source of increased pH seen in the mid 1960s. Wang (1984) found that one wildfire replication studied created a loss of N in soil that would take approximately 30 years to replenish via natural atmospheric N deposition.

Methods of evaluating water quality and disturbance

To determine spatial aspects of land cover on water quality trends in a South Korea basin, Chang (2008) used stepwise multiple linear regressions with seasonal Mann-Kendall trend statistics and the Moran's I test for spatial autocorrelation. His findings demonstrate the importance of scale and area of analysis on results with water quality trends differing across the region and depending on point-source or non-point source perturbations. The USGS conducted a study on Toms River tributaries (in the NJ Pinelands) using five years of nutrient and flow data to determine how land-use was related to nutrient loads; separating these data dependent on base-flow or storm flow conditions (Baker and Hunchak-Kariouk 2007). They found that total nitrogen (TN) was lowest in the undeveloped watershed and ammonia was highest in the developed watershed. They studied four watersheds of differing size and development but did not take into account spatial or serial autocorrelation in their analysis. More recently, Tu and Xia (2008) used geographically weighted regression (Fotheringham et al. 2002) to analyze the relationship between land use and water quality. Incorporating spatial relationships on seasonal Mann-Kendall trends improved predictive abilities for their western Massachusetts water quality dataset and indicated that SC was the best indicator for predicting disturbance. King (2005) demonstrated that landcover classes are generally collinear and spatially autocorrelated; therefore explicit spatial considerations must be utilized in understanding the relationships between water quality measures and disturbance. They used distance weighting of landcover classes to improve predictions of N loads in small watersheds.

Methods of impact assessment, including the before/after control/impact (BACI) method, (which involves concerns ranging from irregular measurements to changing habitats in control versus impact sites (Smith et al. 1993)), are used to understand whether disturbances create significant changes to ecosystem variables. This method has been championed by Stewart-Oaten (1986) but has been criticized by Underwood (1994) and others for its lack of power due to lack of control site replications and the knowledge needed to assess natural spatial and temporal variation in comparison to impact sites. More recently, Johnson (2005) introduced a disturbance index meant to overcome some of the assumptions needed to analyze data under the BACI framework. This index is not comparable between parameters, but can be used to determine if an impact is outside of normal variation (plus or minus two standard deviations) for a particular parameter. The standard deviations are calculated from the pre-disturbance data only.

Aspects of water quality data

Flow is a large part of any hydrological study. In the Pinelands, Zampella et al’s (2001) study found that long term monitoring sites provided a good estimate of flow for those sites nearby with short term records. Whether researchers flow-adjust parameters can depend on whether the parameter being measured originates from a point or non-point source and parameters can have linear, log or inverse relationships to flow (Cavanaugh and Mitsch 1989). Potential trends were successfully addressed by using flow weighted water quality parameters in a before/after control/impact (BACI) study on the effects of a power plant installation on fish populations (Smith et al. 1993).

Trend detection in water quality data has been a pivotal issue in the hydrology literature for some time. Problems with understanding trends in water data include non-normal distributions, seasonality, flow, missing values, values below detection levels and serial correlation (Hirsch et al. 1982). Many studies have shown that fire affects water quality over short to long time periods and can differ according to numerous environmental factors (Forman and Boerner 1981; Minshall et al. 2001). Researchers have also addressed the importance of incorporating serial (temporal) and spatial correlation (Hirsch et al. 1982; Khaliq et al. 2009). Clement and Thas (2007) discussed a non-parametric spatio-temporal trend detection method for river monitoring networks where spatial dependence is based on flow direction within the network. These concerns may largely be outside of the realm of analysis for datasets with only a few temporally sparse water quality measurements before and after a disturbance (e.g. a fire date) or over separate and discrete areas for each finite disturbance (e.g. fire perimeters).

Overall, there are many challenges to successfully conduct a landscape level/long term study comparing fire to water quality parameters; however, we feel that by using tools from hydrologists combined with landscape ecologists, we can arrive at a better understanding of how water quality is affected by fire in comparison to other disturbance events (such as LULC change). We may also be able to refine the data requirements needed to improve our ability to detect fire impacts.

Methods

Study Area and Data Sources

The area of investigation included the Barnegat Bay and Mullica River watersheds, the two watersheds upstream from the JCNERR estuaries (Figure 1). Large fires (>40.47 ha/100 acres) falling within, or partially within either of the two watersheds were selected for analysis (La Puma 2011, under review). Surface water quality measurements including pH, specific conductance, turbidity along with flow rates from the United States Geological Service's (USGS) National Water Information System (NWIS) and the Environmental Protection Agency's (EPA) data system, STORET, were selected using a series of queries in Microsoft Access (Microsoft Corporation 2010). Queries filtered water quality records for all possible samples containing a before and after reading corresponding to available fire dates. Out of ~55,000 water quality records and approximately 500 large fires (1924-2007) in our watersheds, fires with before/after water quality measurements within 30 days of the start date of the fire consisted of these parameters: pH (n=11 fires); specific conductance (SC, n=12 fires), turbidity (n=14 fires). Nutrient data was sparse and therefore excluded from this study. Additionally, non-fire controls having the same day and month of the fire date (but different year) were chosen based on availability of a non-fire year water quality samples in years preceding the fire. If non-fire year data was not available for preceding years, the most recent available dates of sampling in the water quality record were used to minimize any affect of fire on non-fire year data. For example, if a fire occurred in 1982 and if non-fire data were not available for years preceding the fire, but the available water quality record extended to 1986, data from 1986 would be used for non-fire year values as the most temporally distant from the fire date itself. A maximum of 34 days before/after the fire date equivalent (day and month) of the non-fire year were accepted for analysis which resulted in matched pairs of fire year/ non-fire year data for ph (n=10), SC (n=11) and turbidity (n=7). Days before and after a fire date a water sample was taken for fire and non-fire years were calculated in Microsoft Access (Microsoft Corporation 2010).

Flow data were obtained from two long term USGS monitoring stations, one in the Mullica River watershed (site: 1410000, Oswego River) and one in the Barnegat Bay watershed (site: 1408500, Toms River). Flow from these long term stations was used as a proxy for flow at all water quality stations within the respective watershed due to the fact that most stations had sporadic data availability for flow, whereas the USGS stations had consistent flow information for over 70 years. Flow on the date water samples were collected was used to flow-adjust parameters having a detectable correlation with flow.

Geographic information systems (GIS) analyses

For comparison to water quality parameters several spatial indicators were calculated via GIS. The percent of land burned within the basin area contributing to the water quality station was calculated via cumulative watershed area attribute data associated with each sub-watershed. If stations were not located at the mouth of sub-watersheds (common) but were along a stream within the sub-watershed, the areas to be added or subtracted from the cumulative sub-watershed areas were calculated via topographic estimations. Total hectares burned for fire perimeters were calculated via XToolsPro (XToolsPro 2003-2008) in ArcGIS10 (ESRI 2011). The stream distance from the stream closest to the downstream edge of the fire perimeter to the monitoring station where the data was collected was calculated using the cost-distance origin/destination function of Network Analyst in ArcGIS10 (ESRI 2011). This function is specifically designed to use distance attributes associated with network segments (such as parts of streams) to find the shortest distance via the network to a destination (e.g. the sampling station).

Statistical analyses

Although numerous statistical techniques were considered, including paired t-tests, Wilcoxon rank tests, seasonal Mann-Kendall trend analyses, geographically weighted regressions, BACI, and the disturbance index; data availability limited the analysis to Wilcoxon rank tests, BACI, ANCOVA, multiple regression and disturbance index tests. Five years of water data is needed to perform the seasonal Mann-Kendall trend analyses before and/or after a fire and this was not available for any fires in our sample. Spatial analyses were also not available due to differing fire dates. If data had been concurrent (e.g. many fires occurring on the same date with accompanying water station data) then spatial regression techniques would be appropriate. Additionally, if several fires occurred in the same location, serial correlations may have been a concern; however, those fires which had corresponding water data did not occur in the same locations as previous fire/water quality data.

Spearman’s rank correlation was used for investigations of relationships with flow and pH, SC and turbidity data from before/after fire and non-fire year data (i.e. all of our data).

The paired Wilcoxon rank test was used to test for differences between before and after fire data for flow-adjusted pH, raw SC, and raw turbidity values. Additionally, the paired Wilcoxon rank test was used to compare fire and non-fire year difference magnitudes. Differences were computed by subtracting the *after* fire date parameter values minus the *before* fire date parameters (after non-fire year minus before non-fire year vs. after fire year minus before fire year, see Table 1).

BACI and disturbance index tests were also performed to assess the impact of fire on water quality data. For the BACI, flow-adjusted pH, raw SC and raw turbidity *before* fire date parameter differences were tested against *after* fire date parameter differences using the non-parametric paired Wilcoxon rank test (Stewart-Oaten et al. 1986), a slight modification of the methods used above (before fire year minus before non-fire year vs. after fire year minus after non-fire year). Boxplots displaying the data compared in each paired Wilcoxon rank test above were also created to visualize data distributions.

The disturbance index for pre-fire (before) data (PreFDI) was calculated as follows (Johnson et al. 2005):

$$\mathrm{Pre}FDI = \frac{(Fy / NFy)}{1/|(Fy - NFy)| + 1}$$

where *Fy* is simply data from a fire year and *NFy* is data from a non-fire year. Subsequently, the mean and ± 2 standard deviations (SD) were calculated across all samples from the pre-fire disturbance indices. The post-fire (PostFDI) disturbance index was calculated using post-fire data. If the post-fire disturbance index fell outside of the ± 2 SD range, then the disturbance for any particular sample date was considered significant. Additionally, once the DI was calculated all fires were ranked according to their disturbance level for each parameter.

Analysis of covariance was used to test the before minus after fire date difference in water quality parameters (all difference data were normal except for SC) against fire year (Y/N) as a factor and distance to station (network stream distance), percent of basin burned (based on total area draining to the water station), days before fire date water samples were taken, and days after fire date water samples were taken as additional predictor variables (fire date was the fire start date). Additionally, multiple regression was used for fire year parameters alone. Once the full model was assessed, a StepAIC procedure was used to find the best predictor variables from the full model for each parameter (Venables and Ripley 2002). All analyses were performed using the R statistical package (R Development Core Team 2009).

Results

Due to the restrictions we posed in terms of timing and spatial location, only a small sample size of study basins and fire events was deemed suitable. As stated above, out of ~55,000 water quality records and approximately 500 large fires (1924-2007) in our watersheds, fires with before/after water quality measurements within 30 days of the start date of the fire consisted of these parameters: pH (n=11 fires); specific conductance (SC, n=12 fires), turbidity (n=14 fires). The percent area of the basin upstream of fire that was burned ranged from less than one percent to 9.886% (see Table 3).

For our first step of analyzing relationships between water quality and fire, we determined if flow adjustment was important for any of our parameters. Using the Spearman's rank correlation for non-normal data we found that pH was inversely related to flow in our sample ($p = .003$, see Figure 2), but SC and turbidity were not related to flow.

Median differences between before and after fire values for flow-adjusted pH, raw SC and raw turbidity were tested using the paired Wilcoxon rank non-parametric test due to non-normal distributions (tested via the Shapiro-Wilk's test for normality after transformations for normality were unsuccessful). Although boxplots showed slight increases in all parameters after fire, no significant differences were found for pH, SC, or turbidity (see Table 2 After fire vs. before fire and Figure3). When adding in non-fire years as controls, we again found that boxplots of the data showed slight gains for fire years as compared to non-fire years, but we found no significant differences in the medians of after minus before data for fire vs. non-fire years using the Wilcoxon rank test (again, distributions were non-normal, see Table 2 and Figure 4). In general outliers were not removed for statistical tests because of the small sample size although we did remove the two largest fire and non-fire year raw data outliers to better display the SC data in the fire year vs. non-fire year and BACI (see below) SC boxplots only. Using the BACI methodology, pH, SC and turbidity showed no significant results using a paired Wilcoxon rank test on the pre versus post-fire differences (before fire year – before non-fire year vs. after fire year – after non-fire year) and results were identical to the fire year vs. non-fire year differences above (Table 2, Figure 5).

The disturbance index results indicated that two fires were outside of the $\pm$ 2 SD range of PreSDI data, with the 1977B072702 fire significantly affecting flow adjusted pH and the 1977B051403 fire significantly affecting turbidity levels (Table 3). In both cases, the parameter increased.

Analysis of covariance with the main factor of fire year, and the explanatory variables of: computed stream network distance from station to fire perimeter, the percent area burned within the station's cumulative basin area, hectares of the fire perimeter, and the number of days before and after a fire date the water sample was taken; demonstrated that whether or not a fire occurred was not significant for any parameter using the after minus before fire difference as the dependent variable. From this result, we restricted our linear analysis to fire year data only to explore the influence of the spatial and temporal variables. Differences in (after minus before) fire values displayed normal distributions after transformation for pH (square root of flow adjusted pH, raw turbidity). We were unsuccessful at transforming SC for normality using log transform or square root transformation methods. No explanatory variables tested were significant for fire year parameters (Table 4). The StepAIC procedure produced the most parsimonious model from all of our explanatory variables (Table 5). For pH, the number of days after a fire was significantly related to (after minus before) fire date samples. The percent of basin burned along with hectares burned improved the model but the overall model was not significant. For turbidity, the best model included the percent of the basin burned and hectares burned but neither was significant. SC was inconclusive with the best model containing only the intercept. The power of this analysis was low with the small sample size involved for each parameter.

Discussion

The landscape perspective of hydrology in relation to wildfire disturbance requires flexibility regarding spatial and temporal aspects of any ecological data gathered (Luce and Rieman 2010). Due to the data sources involved, possibilities of an in depth analyses of wildfire's effects on water quality across the landscape were limited. However, simple tests of significant differences provide a beginning point in understanding which water quality parameters it may be necessary to monitor more closely in order to understand the extent of ecosystem services provided by or limited by specific fire management plans. Even with limited landscape level data, conclusions can be drawn that can help in designing more detailed landscape level studies.

After a fire, the medians of pH, SC and turbidity rose slightly but were not significantly different than before fire values at the .05 significance level. It is not atypical to see a rise in pH, SC and turbidity after wildfire (National Wildfire Coordinating Group 2001), therefore the Pinelands may be following general ecosystem trends, even with low nutrient levels and acidic standard conditions. These results are in line with the results from other wildfire studies when accounting for the effects of increased ash deposits on specific conductance and turbidity and increased cation exchange for pH. When comparing the magnitude of the difference for fire years versus non-fire years, fire years show more positive gains than non-fire years for pH, SC, and turbidity (meaning that these parameters increased more after the day and month specified for a fire year than they did for a non-fire year). Additionally, the range of data as indicated by the boxplots increased for turbidity in fire years as compared to non-fire years. The BACI test indicated that when using non-fire years as controls, these data contained too much variability to detect significant differences between control and impact data. Although most BACI studies use nearby or similar sites for control data, in the case of fire, using the same site for controls in previous non-fire years seemed to be the best comparison in our system. However, some sites did not have non-fire years before the fire date, therefore using following years at the same site for non-fire years may have confounded the water quality results. As mentioned, many studies indicated that there may be long term recovery times for water quality measures, potentially compromising these data as true controls using the BACI method (Boerner and Forman 1982).

The fire year vs non-fire year test (after fire year – before fire year vs. after non-fire year – before non-fire year) should indicate whether the magnitude of change from disturbance years was significantly different from non-disturbance years. The BACI test (before fire year – before non-fire year vs. after fire year – after non-fire year) should indicate whether seasonal differences are significantly different from disturbance differences. The fact that the Wilcoxon rank results for both categories of tests above produced the exact same results for all three parameters tells us that seasonal differences cannot be distinguished from differences that come about as a result of wildfire disturbance. The disturbance index we calculated also showed significance for two fires, but there was no pattern to the ranking of spatial variables for these two fires based on their disturbance index rank. More samples may have elucidated a spatial relationship with the disturbance index.

Issues related to significance testing indicate that water quality sampling in the Barnegat and Mullica River watersheds is insufficient to determine temporal intensity and geographic extensiveness of the effects of disturbance in the form of fire. Due to the sporadic and unpredictable nature of fire, it would be necessary to undertake monitoring at long term gauging stations at regular intervals (and perhaps at higher intervals during peak fire season, e.g. April) to obtain enough data for a robust understanding of the effects of fire on water quality measures. Also, determining trends before and after a fire would add another level of understanding to the dynamics of the ecosystem and the intensity of the disturbance. Disturbances such as landcover change are less fleeting in nature and therefore studies on this type of disturbance in the Pinelands has typically been enumerated for those times when water quality data is abundant and/or can be diligently collected for a few years independently. Wildfire is unpredictable and pre-fire data is of utmost importance in determining its effects.

The linear model did not show significance for fire vs. non-fire years for pH, SC, and turbidity which points to the necessity for more information on these important disturbance indicators. When using our model refinement procedure of StepAIC, the only significant explanatory variable out of all of our tests was the number of days after a fire a sample was taken for pH. Interestingly, the fire date from which the number of days after a fire a sample was taken was calculated from was the start of the fire, not the end. The beginning date of the fire was used as a necessity, since records only included ‘under control’ dates and ‘fire out’ dates. ‘Under control’ dates could mean that a fire would burn many more acres, but the fire managers felt it would be contained. ‘Fire out’ dates meant that all possible hotspots were squelched, but that the main fire could have been out for some time before that. As such, there was no method of ascertaining whether a fire had burned through most of it’s final perimeter for either of those two dates; however, start dates had no such stipulations but could be confounding the ‘after fire’ water quality readings if the fire were still burning at the time the parameter was sampled, or if the fire was out on the same day it started. In essence this creates a system where the date a water quality parameter was taken after a fire start date becomes very important as to how effective it is at measuring the effects of the disturbance.

A rise in pH, SC, and turbidity would not be beneficial to the Barnegat Bay in the JCNERR, already battling eutrophic conditions; however, the relative effect of wildfire versus other causes of eutrophication needs to be addressed. For example, the slight rises in pH and SC we saw from fire had medians of 5.15 for pH and 50 µmhos/cm. When compared to Zampella et al (2007b) our medians from wildfire disturbance correspond to the 35% watershed disturbance level along a gradient of median readings for watershed disturbance within sub-watersheds in the Mullica River Basin (<10% disturbed was considered pristine). In essence, the shorter term pulses of change induced from wildfire we saw, if shown to be significant in future studies, represents only a fraction of the changes that have been shown to be caused by altered land in the pinelands system with medians of up to 7.0 for pH and 125 µmhos/cm for SC for the highest levels of watershed disturbance due to urban and agricultural land use. The long term rises in pH and SC seen with altered land were tied to increases in nutrient levels as well (Zampella et al. 2007b).

While the best available evidence we have does not show wildfire have a major impact on Pinelands water quality, we do not have great confidence in the sufficiency of these available data source to adequately determine whether fire affects water quality in our system on the landscape level. Even though these increases seem to be inconsequential in the larger disturbance spectrum, more study is needed to conclusively state that these slight increases are due to wildfire and not other factors such as seasonality trends or concurrent LULC change. Additionally, the greatest percentage of basin burned for our study was <10%. A larger sample size along with a higher percent of watershed basin area burned is needed to more fully elucidate the effect of wildland fire and downstream water quality.

Other hypotheses needing further investigation include a seasonal effect in that during the Spring burning season, growth and nutrient uptake is at it's highest in the ecosystem, potentially absorbing any flush of nutrients becoming available from fire. Additionally, fine clay soils underlying the pine plains have been hypothesized to capture much of the nutrient runoff from fire and may be excellent at capturing nutrients needed for pitch pine re-growth (Walter Bien, Drexel University, personal communication). If surface runoff is not the dominant input into streams, some nutrients may be diluted from mixing with groundwater, masking an effect of fire. Further research should concentrate on how pulses of pH, SC, and turbidity may contribute to downstream ecosystem conditions and how these parameters might change over time. A targeted study, rather than an exploratory data mining study such as ours is needed to properly address the question of the affect of wildfire on water quality, but this type of study will not be possible until the limitations of the current datasets such as sporadic data collection, lack of flow data and limited nutrient samples are improved. In addition, the water quality monitoring network of the Mullica and Barnegat Bay watersheds should be enhanced (especially temporally after fire) in order to elucidate the full effects of wildfire on pinelands streams and the JCNERR estuaries.

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Table 1. Example of water data collection dates in relation to a specific fire date. Dates before and after a fire varied depending on when water samples were collected. Tests listed are associated with the corresponding rows and columns of data (for all fire and water data available) with the figures that correspond to those tests.

	Sample collection date Before Fire	Fire Date	Sample collection date After Fire	Tests: Figure 3	Tests: Figure 4
Fire year (Fy)	3/25/1973	4/20/1973	4/29/1973	FyAfter vs. FyBefore	FyAfter - FyBefore vs.
Non-fire year (NFy)	3/29/1972	4/20/1972	4/25/1972		NFyAfter - NFyBefore
Test		Figure 5			
BACI	FyBefore-NFyBefore	vs.	FyAfter-NFyAfter		

Table 2. Paired Wilcoxon's rank on pH, SC and turbidity.

Tests are for after (FyA) vs. before (FyB) fire; after-before for a fire year (FyA-FyB) vs. after-before for a non-fire year (NFyA-NFyB); and the before after control impact (BACI), pre-fire: before (FyB) – before (NFyB) vs. post-fire: after (FyA) – after (NFyA).

After fire vs. before fire		
Parameter	Transformation	p-value
pH	flow-weighted	0.240
SC	none	0.806
Turbidity	none	0.674

Fire year difference vs. non-fire year difference (After - Before)		
Parameter	Transformation	p-value
pH	flow-weighted	0.432
SC	none	0.758
Turbidity	none	0.735

BACI: Before fire (Fy) - Before fire (NFy) vs. After fire (Fy) - After fire (Nfy)		
Parameter	Transformation	p-value
pH	flow-weighted	0.432
SC	none	0.758
Turbidity	none	0.735

Table 3. Ranked disturbance index values for each fire with water quality data downstream. Asterisks indicate those fires with parameters outside of two standard deviations of the mean of non-fire year variation.

Rank	FireID	Parameter	Fire Size (ha)	Percent basin burned	Distance to station (km)	Disturbance Index (DI)
1	1977B072702	pH/flow	440.437	4.051	8.219	*2.055
2	1980B051402	pH/flow	49.688	0.094	38.510	1.001
3	1981B041808	pH/flow	1384.764	9.886	7.942	0.752
4	1977B051403	pH/flow	377.256	2.263	0.094	0.564
5	1977B033105	pH/flow	455.509	0.093	1.706	0.504
6	1980B051707	pH/flow	49.688	1.113	6.363	0.299
7	1982B072201	pH/flow	376.964	0.599	38.248	0.281
8	1977B041804	pH/flow	307.184	2.673	0.000	0.047
9	1979B050806	pH/flow	76.750	1.051	8.487	-0.007
10	1978B040104	pH/flow	168.745	1.186	0.000	-0.899
1	1977B041804	SC	307.184	2.673	0.000	1.215
2	1977B033105	SC	455.509	0.093	1.706	1.118
3	1980B051402	SC	49.688	0.094	38.510	1.076
4	1983B073001	SC	616.150	5.709	8.110	1.048
5	1979B050806	SC	76.750	1.051	8.487	0.980
6	1977B051403	SC	377.256	2.263	0.094	0.894
7	1977B072702	SC	440.437	4.051	8.219	0.706
8	1980B051707	SC	49.688	1.113	6.363	0.694
9	1978B040104	SC	168.745	1.186	0.000	0.508
10	1982B072201	SC	376.964	0.599	38.248	0.373
11	1981C111201	SC	213.459	0.339	13.804	0.119
1	1977B051403	Turbidity	377.256	2.263	8.487	*12.727
2	1979B050806	Turbidity	76.750	1.051	0.000	3.000
3	1980B051402	Turbidity	796.117	0.094	6.363	1.296
4	1980B051707	Turbidity	49.688	1.113	0.000	1.050
5	1977B041804	Turbidity	307.184	2.673	0.094	0.583
6	1981B041808	Turbidity	1384.764	9.886	7.942	0.278
7	1978B040104	Turbidity	168.745	1.186	38.510	0.208

Table 4. Full model analysis of covariance for fire year data.

pH	p-value
Distance to station	0.347
Percent basin burned	0.248
Days before fire	0.359
Days after fire	0.379
Hectares burned	0.314
Overall model	0.251
SC	
Distance to station	0.925
Percent basin burned	0.861
Days before fire	0.490
Days after fire	0.439
Hectares burned	0.798
Overall model	0.890
Turbidity	
Distance to station	0.759
Percent basin burned	0.202
Days before fire	0.396
Days after fire	0.564
Hectares burned	0.158
Overall model	0.599

Table 5. Best fit variables for linear model determined by StepAIC for pH (df = 3 and 7) and turbidity (df = 2 and 11).

stepAIC

pH	p-value	
Percent basin burned	0.086	
Days after fire	0.027	*
Hectares burned	0.113	
Turbidity	**p-value**	
Percent basin burned	0.122	
Hectares burned	0.111	

* indicates significance at the .05 level

Figure Captions:

Figure 1. Locations of fires with the closest downstream water stations having data pertaining to those fire dates (e.g. before and after a fire date) as well as the USGS flow reference sites.

Figure 2. Scatterplots of a) pH, b) SC, and c) turbidity in comparison to mean flow on the date of sample collection. Data were non-normal and tested via Spearman's rank correlations.

Figure 3. Within a fire year (FyBefore vs. FyAfter) parameters showing values tested via the paired Wilcoxon's rank test for a) pH, b) specific conductance and c) turbidity.

Figure 4. Fire year (FyAfter minus FyBefore) versus non fire year (NFyAfter minus NFyBefore) difference values used in the paired Wilcoxon's rank test for a) pH, b) specific conductance and c) turbidity. Positive values indicate that the parameter increased after fire date. Negative values indicate that the parameter decreased after fire date.

Figure 5. Boxplot of pre-fire and post-fire difference results used for the before/after control/impact (BACI) paired Wilcoxon's rank rest for a) pH, b) specific conductance and c) turbidity (FyBefore - NFyBefore vs. FyAfter - NFyAfter). Positive values indicate that the parameter was higher in a Fy than in a NFy. Negative values indicate that the parameter was lower in a Fy than in a NFy. Additionally, pre-fire data show the range of variation across seasons regardless of whether a fire occurred.

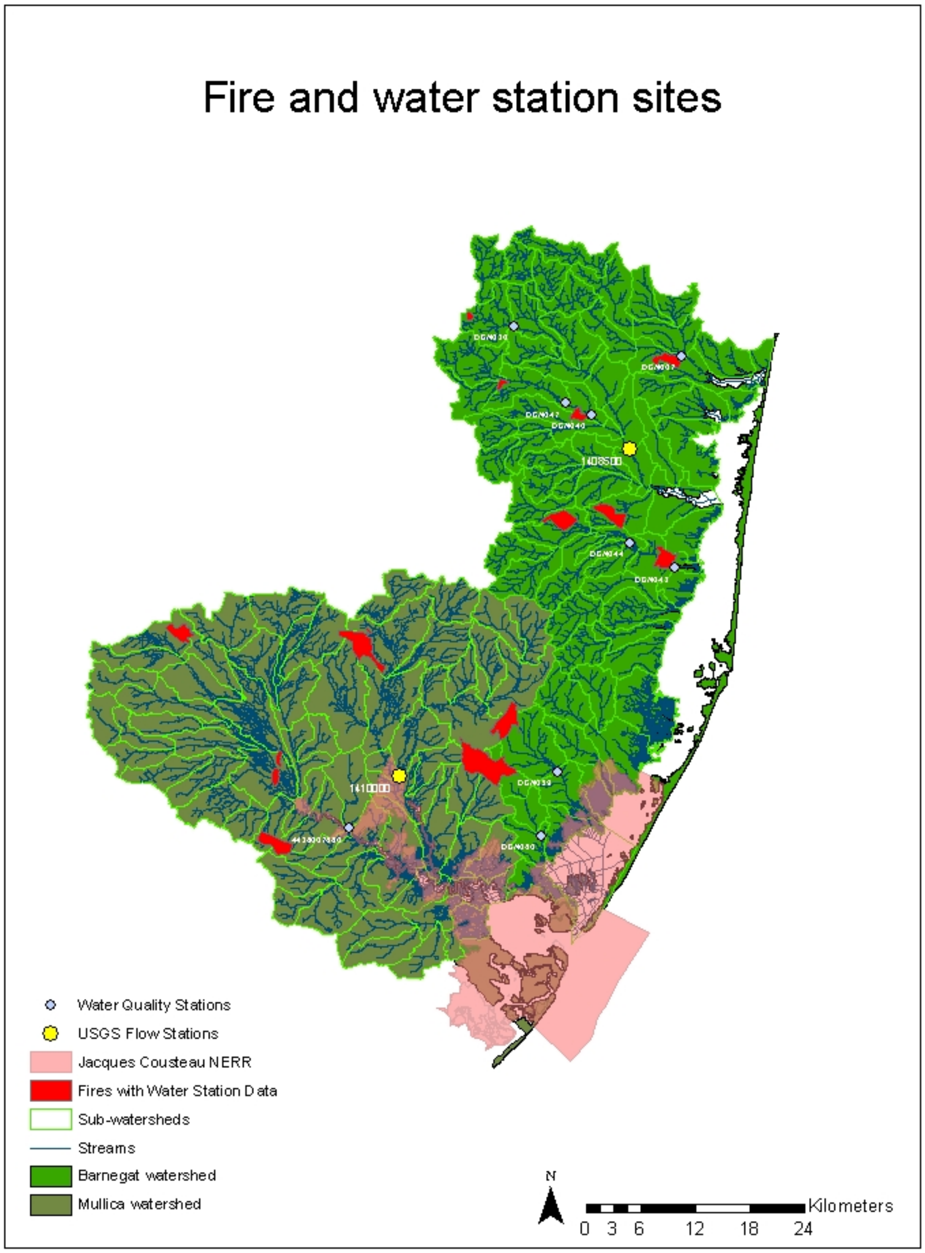
Fire and water station sites
Water Quality Stations
USGS Flow Stations
Jacques Cousteau NERR
Fires with Water Station Data
Sub-watersheds
Streams
Barnegat watershed
Mullica watershed
N
Kilometers
0 3 6 12 18 24

Figure 1.

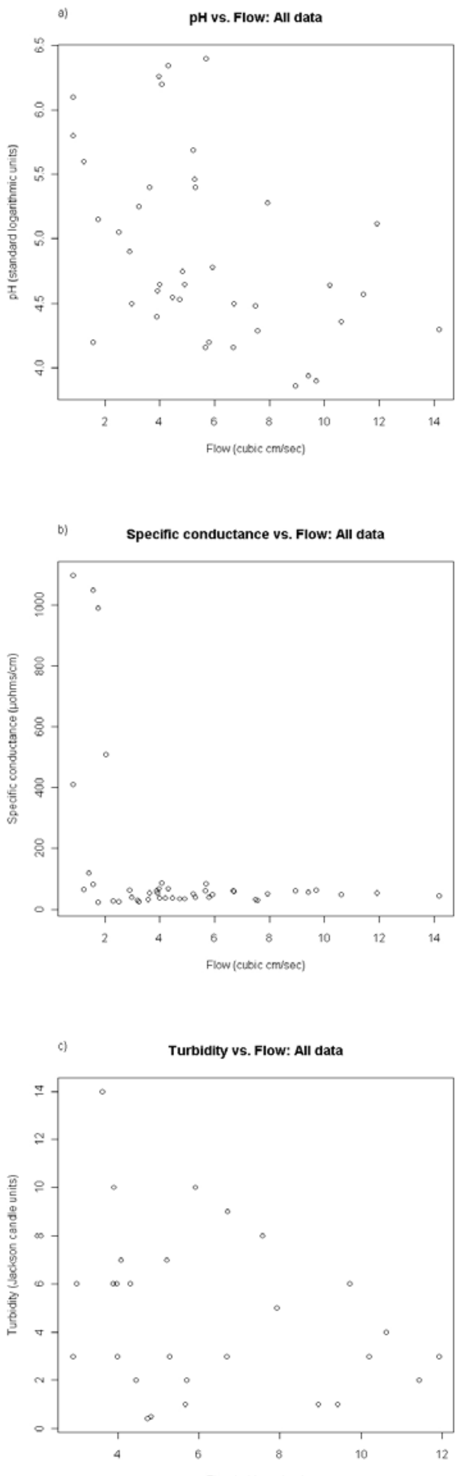
a)
pH vs. Flow: All data
pH (standard logarithmic units)
Flow (cubic cm/sec)
b)
Specific conductance vs. Flow: All data
Specific conductance (µohms/cm)
Flow (cubic cm/sec)
c)
Turbidity vs. Flow: All data
Turbidity (Jackson candle units)
Flow (cubic cm/sec)

Figure 2.

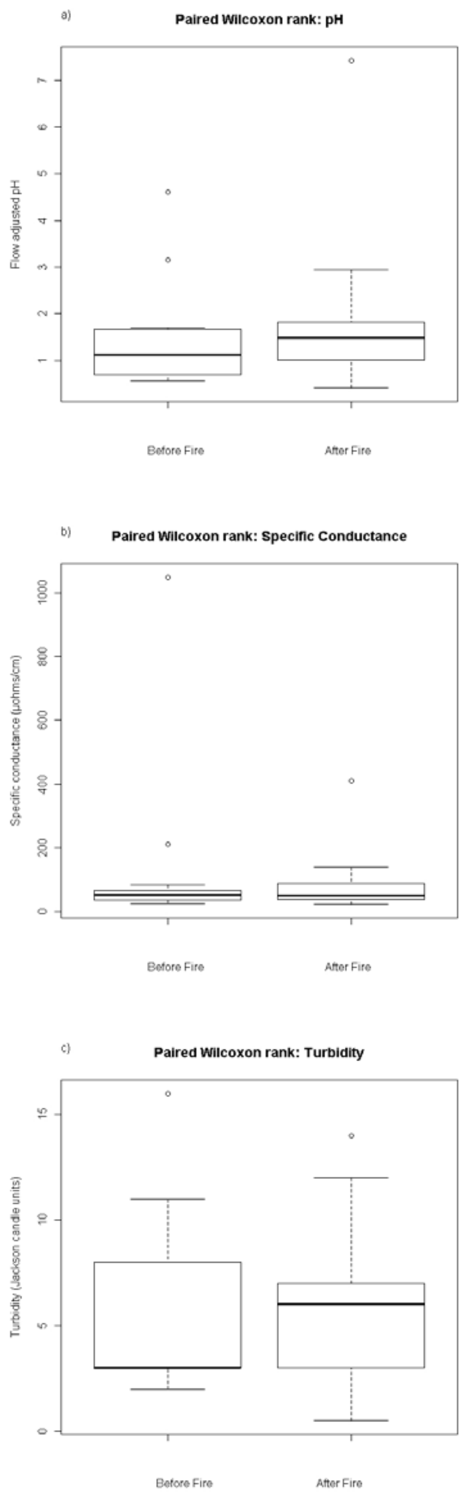
a)
Paired Wilcoxon rank: pH
Flow adjusted pH
1
2
3
4
5
6
7
Before Fire
After Fire
b)
Paired Wilcoxon rank: Specific Conductance
Specific conductance (µohms/cm)
0
200
400
600
800
1000
Before Fire
After Fire
c)
Paired Wilcoxon rank: Turbidity
Turbidity (Jackson candle units)
0
5
10
15
Before Fire
After Fire

Figure 3.

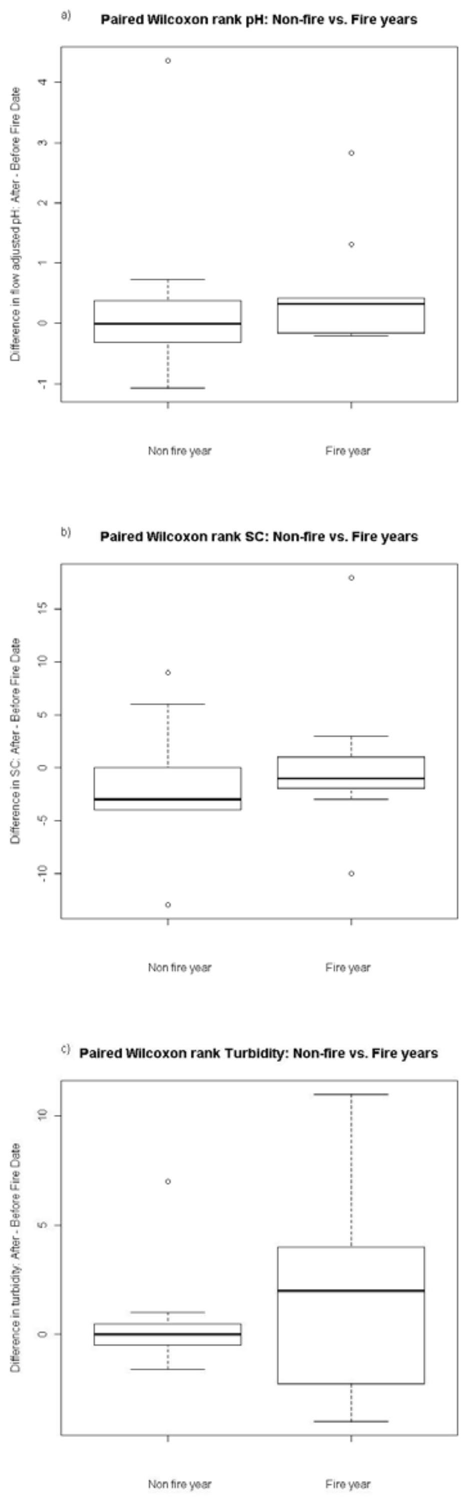
a)
Paired Wilcoxon rank pH: Non-fire vs. Fire years
Difference in flow adjusted pH: After - Before Fire Date
-1
0
1
2
3
4
Non fire year
Fire year
b)
Paired Wilcoxon rank SC: Non-fire vs. Fire years
Difference in SC: After - Before Fire Date
-10
-5
0
5
10
15
Non fire year
Fire year
c)
Paired Wilcoxon rank Turbidity: Non-fire vs. Fire years
Difference in turbidity: After - Before Fire Date
0
5
10
Non fire year
Fire year

Figure 4.

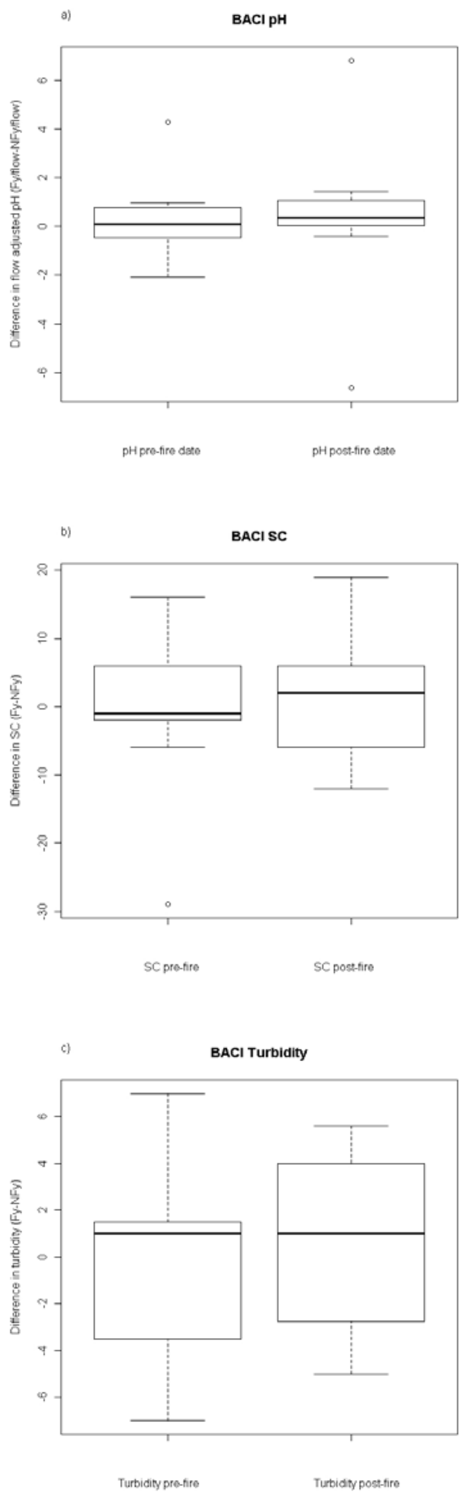
a)
BACI pH
Difference in flow adjusted pH (Fy/flow-NFy/flow)
6
4
2
0
-2
-4
-6
pH pre-fire date
pH post-fire date
b)
BACI SC
Difference in SC (Fy-NFy)
20
10
0
-10
-20
-30
SC pre-fire
SC post-fire
c)
BACI Turbidity
Difference in turbidity (Fy-NFy)
6
4
2
0
-2
-4
-6
Turbidity pre-fire
Turbidity post-fire

Figure 5.

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CHAPTER 3

Landscape disturbance and succession modeling in the Pinelands of New Jersey using LANDIS-II: The implications of human influence on fire and forest composition

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Abstract:

Coupled human-natural systems present complex relationships between human influence and ecosystem response and services. The Pinelands of New Jersey represent a highly human influenced system noted for its fire regime which helps maintain pinelands cover and halt succession from pine to oak forest composition. The goal of this research was to use a landscape disturbance and succession model (LANDIS-II) to understand long term implications (over 100 years) of human altered land on fire regimes and thus pine versus oak forest cover. Previous research (Chapter 1) showed that fire frequency in the ecological wildland urban interface (EWUI) is depressed due to fire suppression and that these areas are transitioning from pine to oak forest composition at a faster rate. We modeled current altered land and future maximum build-out and added EWUI fire regimes for both of these scenarios to determine the extent of shifts for pine and oak forest cover under a modern fire regime. Increased fragmentation due to both the build-out scenario and added EWUI no-fire zones decreased average fire sizes and increased pine to oak transitions. LANDIS-II forecasts provide a range of possible future scenarios and a decision-support tool for understanding how land-use and fire management policies may affect ecosystem processes and patterns.

Introduction

Coupled human-natural systems have become a focus in landscape ecology and are influenced by factors such as government policies, ecosystem context, and ecosystem services (Liu et al. 2007). Temporal and spatial thresholds of these highly diverse factors can determine the rates at which ecosystems transition and/or fundamentally change (Liu et al. 2007). Key research areas for human-natural systems include modeling of thresholds with different scenarios of human management. The Millennium Ecosystem Assessment (2005) was initiated in 2001 by Kofi Annan of the United Nations to gain insight into spatial and temporal limits to ecosystem services subject to human impacts. Their findings suggest that humans have changed ecosystems more rapidly in the past 50 years than any time in human history, potentially permanently altering ecosystem services such as clean water, food and other natural resources (Millenium Assessment 2005). The coupled human-natural system of the New Jersey Pinelands represents a complex spatial, social and ecological relationship between human settlement and its effects on the fire regime and forest composition.

Spatial, social and ecological influences in the Pinelands

A long history of human-caused fire (via railroads during the late 19th century, a bombing range during the late 20th century, and continuous accidental and incendiary ignitions) has essentially sustained large contiguous tracts of pine dominated forest (Forman and Boerner 1981); perhaps with more pine cover than pre-colonial composition, when oak cover may have been more prominent with less frequent human-caused ignitions (Kurczewski and Boyle 2000; Oswald et al. 2010; Watts 1979). Human land-use decisions also play a major role in the configuration of natural forest lands in relation to human development. The Pinelands Commission, an entity created in 1978 and charged with protecting the cultural and environmental integrity of the NJ Pinelands National Preserve, has successfully slowed development in core natural forest areas (Hasse and Lathrop 2001). However, there are still areas of the NJ Pinelands in which different levels of development are allowed based on the Pinelands Commission Comprehensive Management Plan (CMP), and these areas have steadily become more contiguous due to added development since the CMP was implemented. Forecasts of maximum build-out, or scenarios where all developable land is developed, show that with current rules the potential for continued growth is substantial (Lathrop et al. 2003; Lathrop and Conway 2001). From the social perspective, issues such as fire safety and development restrictions as well as available funding at the state and local levels have influenced the landscape of the NJ Pinelands today. The New Jersey Forest Fire Service's (NJFFS) mandate since its founding has been to suppress all fires for public safety reasons, and this policy has led to a build up of fuels in areas not controlled by prescribed burning. Historically, lack of staff and state funding has prevented any modification of this perspective. However, recently, fuel treatments have been funded through national sources to reduce fuels by thinning in the wildland-urban interface (WUI) at key defensive points (Mike Drake, New Jersey Forest Fire Supervisor, personal communication). This thinning has the added service of emulating low intensity fire by reducing ladder fuels (http://www.nj.gov/dep/parksandforests/fire/whm-planning.htm).

The effects of a changing fire regime, either increased or decreased, are far reaching for pinelands ecosystems. Most pinelands ecosystems require fire in order to persist (Gilliam and Platt 2006; Glitzenstein et al. 2003; Hofstetter 1984; Loudermilk et al. 2011; Vaillant and Stephens 2009); although see Johnstone et al. (2010) for a case of deciduous dominance after severe fire in Alaska. Fire frequency and severity can determine succession trajectories of forest composition but can also influence understory vegetation, habitat (Keith et al. 2010), and microenvironment (Nowacki and Abrams 2008); providing positive feedbacks to non-pine succession. Lack of fire generally leads to a build up of ladder fuels in the form of understory growth (Clark et al. 2009b; Skowronski et al. 2007) and gradual succession to oak dominated forests (Gunderson 1994; Hofstetter 1984). In the Pinelands of New Jersey, a clear relationship between fire frequency and its necessity for pine persistence has long been recognized and is similar to Long Island Pine Barrens and New England coastal pine ecosystems (Forman and Boerner 1981; Parshall and Foster 2002; Parshall et al. 2003; Windisch 1999, also see Chapter 1).

Forest composition, in turn, determines carbon sequestration, soil composition and water filtration processes within the Pinelands ecosystem. Pine versus oak dominated forest is directly related to fire frequency in the Pinelands (see Chapter 1) which is determined by adjacency to altered land. Whether clumped or interspersed in the natural landscape, current and future patterns of human altered land and policies determine where and how the historic fire regime (whether human caused or naturally induced) can either continue on a similar track, or head towards a new trajectory. As the modern human footprint and its physical and biological impacts on the landscape become established we can start to understand implications for long term ecological change.

Previously (Chapter 1), I defined an area termed the ecological wildland urban interface (EWUI), or areas adjacent to altered land where the fire regime had drastically declined over the last 80 years. Additionally, those areas that were close to altered land and not experiencing fire showed the greatest percentage of land transitioning from pine to oak dominated forest during the 20 year period studied. In the Pinelands of New Jersey, this type of forest transition and resulting change in fire regime may be intensifying due to the configuration of altered land. The perpetuation of the reduced fire regime experienced in the last few decades may continue or accelerate the rate of succession seen in Chapter 1. Predicting and understanding the future composition of the Pineland ecosystem under different levels of human influence on altered land and fire regimes is the focus of the modeling described in this manuscript.

In order to understand how humans may be affecting long term ecosystem services (such as water quality, fire safety, and carbon sequestration) and in order to assess the implications of human development, fire, and forest change; spatial and temporal forest succession and disturbance modeling efforts are necessary. As yet, modeling of the potential extent of altered land with the added EWUI/ reduced fire effect on fire regimes and change in forest composition has not been investigated either in the Pinelands or elsewhere; therefore, this study represents the first attempt to understand these complex human-environment relations. By modeling the range of future possibilities, we can assess the potential ecological effects of long-term management decisions such as fire suppression and zoning restrictions. Once a range of possibilities is established, we can provide decision support tools for adaptive management given the EWUI effects.

We now have the ability to use vetted and complex technological tools to model how these impacts may evolve or reach thresholds over space and time. Forest landscape dynamic models (FLDM) have evolved from both forestry-based growth-yield and ecologically-based forest succession or physiology models (Mladenoff and He 1999). Spatial versions of Markov or semi-Markov transition models were important precursors to later FLDMs in that they utilized transition probabilities from one timestep to another (Yemshanov and Perera 2002). The combination deterministic/stochastic gap models FORET (Shugart 1984), ZELIG (Urban et al. 1991) and SORTIE (Pacala et al. 1993) all incorporated spatial dimensions to forest succession, but relied on individual trees as modeling units making parameter estimation a large task for any new area of study (Hope 2003). SORTIE also incorporated continuous functions into seed dispersal and competition in areas outside of the immediate neighborhood (Pacala et al. 1993). Continuous functions provide the means to tailor ecosystem effects to impact surrounding areas at different intensities and to variable extents depending on the function used. The MOSAIC model was the first model to connect gap forest succession models (based on extensive empirical parameterization of single plots or stands/ not spatial) and semi-Markov transition models, where interactions and transitions between cells or pixels at each time step are dependent on the history each site (Acevedo et al. 1995). The SELES landscape modeling language has been used for more tailored landscape interactions, however, the user must develop the ecological interactions they wish to model (Fall 1999). The advantages of aggregating gap models to the cohort (or age-based) level and incorporating continuous functions and stochastic functionality for spatial transition probabilities across the landscape were first realized in the LANDIS forest landscape disturbance model (Mladenoff and He 1999). In this manner ecosystem processes have both bottom-up and top-down controls without the imposing amount of detail needed to model at the individual tree or stand level. Additionally, fire and harvest disturbances were added to interact with succession processes with flexible timesteps and spatial scales. LANDIS is tailored to forest disturbance and succession landscape ecology and incorporates current research through its open-source availability, adjustable parameters, and the ability to add extensions through an object-based program.

In this study we used LANDIS-II, a highly regarded second generation landscape disturbance and succession model (Scheller et al. 2007) along with empirically derived spatial data on fire and altered land to investigate the impacts of the human footprint on the fire regime and forest composition in the Pinelands of New Jersey.

LANDIS-II model background

In searching for a spatially distributed community model, LANDIS-II appropriately addressed many of the goals we wished to address in our study. The original LANDIS provided ecosystem-based insights into future forest composition for forests throughout the world (Gustafson et al. 2007; He and Mladenoff 1999; He et al. 2002; Mehta et al. 2004; Scheller and Mladenoff 2004; Scheller and Mladenoff 2005). LANDIS-II, the second generation LANDIS model, is a hierarchical spatial model based on empirical data including species traits such as seed dispersal distances, longevity, age at sexual maturity, shade tolerance, fire tolerance, sprouting abilities, and maximum age of reproduction (Mladenoff and He 1999; Scheller et al. 2007). Inputs include both spatial and non-spatial data sources including: vegetation maps, soil and climate based ecoregions, fire management areas and species establishment probabilities. The landscape level fire disturbance process includes stochastic fire initiations and fire sizes. Fire spread can be constrained by climate, elevation, fuels and management regimes (He and Mladenoff 1999). LANDIS-II also includes modules of insect outbreaks, wind, and harvest as disturbances, although these modules were not included in our study. Current renditions have added biomass as an output using inputs from PnET-II (photosynthesis, evaporation and transpiration) or other physiological model sources for aboveground net primary productivity and species establishment probability inputs. LANDIS-II relies on species age cohorts to track species growth and species traits including shade tolerance, seed dispersal and fire tolerance to estimate competition parameters. Additionally, the model uses empirical data to simulate distance based seed-dispersal for each species and typically removes young or old cohorts given susceptibilities of species to different types of disturbance. Establishment is focused on categorical shade tolerance levels for all species for which seeds are available in neighboring pixels. Adjustable pixel-sized forest communities undergo simultaneous large-scale disturbance and small-scale species competition with neighboring pixels throughout the simulation years, incorporating hierarchical spatial and temporal relationships. Outputs include variable timesteps (up to 500 years) for each species cohort and biomass, as well as fire locations and severity statistics.

In LANDIS-II, fire can be calibrated to mimic actual fire regimes experienced in the study area and succession can be initiated with data extrapolated from field samples of forest composition and age cohorts. However, as a stochastic landscape disturbance and succession model, LANDIS-II is not designed to predict the exact locations of certain forest types in the future. It is designed to estimate landscape level changes based on parameters derived from known species life history traits and effects of disturbance. Once baseline models are established, fire disturbance and other types of disturbance in LANDIS-II disturbances can be adjusted in order to represent future alternative states due to policy changes or management recommendations. Our goal in the study was not to adjust disturbances for our scenarios, but to understand how disturbance would change under different configurations of altered land and with associated fire suppression near altered land.

Methods

Base LANDIS-II Spatial Inputs

For the LANDIS-II model several levels of spatial inputs are required. At the most basic level are ecoregions, or landtypes, representing similar soils and climate. Ecoregions were obtained from the Dynamic Ecosystems and Landscape Lab at Portland State University in Portland, OR and were based on upland and lowland categories from a New Jersey LULC map (Lathrop and Kaplan 2004) as well as soil water holding capacity from the NRSC SSURGO soils dataset (NRCS 2011). These data were generated for a previous study in NJ to assess carbon sequestration in the Pinelands under different fire management scenarios (Scheller et al. 2011) and clipped to our current study area.

Additionally, initial communities across the landscape of all species and cohorts (at 10 yr cohort intervals) are required for initiation of the model. For example, each 100 ha cell is assigned a suite of species and each species is assigned specific age groups (e.g. one pine plains pixel might be assigned two species with several age cohorts such as *Pinus rigida* at 10, 20 and 30, years old and *Quercus marilandica* at 10 years old). This type of cohort based initial community does not take into account the density of trees in each pixel or cell. Scheller et al. (2011) created initial communities using Forest Inventory Analysis (FIA) plots of forest composition and diameter at breast height (dbh) from New Jersey collected between 2005 and 2009 (Hansen et al. 1992; US Forest Service and USDA 2007). Each FIA plot was assessed for a diameter at breast height to age relationship for all plots within the Pinelands and unique species age combinations were identified for each plot (Scheller et al. 2011). Then, each polygon within the landcover map (Lathrop and Kaplan 2004) was assigned a forest type. Also, each FIA plot was assigned a forest type based on dominant species in the plot. From all of the FIA plots associated with each forest type, one was randomly chosen to represent each polygon, resulting in one species combination and age cohort set per forest type polygon (Scheller et al. 2011). This process created a polygon-based initial community map for the entirety of our area of interest based on field data, but extrapolated to represent the entire landscape. Initial conditions in LANDIS-II are considered to be representative of the landscape as a whole and not specific to each pixel. All data are at 100m pixel resolution and in the UTM Zone 18 NAD 83 projection. Spatial extents were clipped to our area of interest in ArcGIS10 using the Extract by Mask tool in the Spatial Analyst extension (ESRI 2011) for the Barnegat and Mullica River watersheds.

Base LANDIS-II parameters

Ecoregions belonged to one of seven different landtypes (Table 1) including: upland with low water holding capacity (ecoregion 2), upland with medium water holding capacity (ecoregion 3), upland with high water holding capacity (ecoregion 4), wetland with low water holding capacity (ecoregion 5), wetland with medium water holding capacity (ecoregion 6), wetland with high water holding capacity (ecoregion 7), and the pine plains area (ecoregion 8) (Scheller et al. 2011). The unique dwarf pine plains area was defined using the area delineated by Givnish (1981) and characterized by a low water holding capacity. Other spatial assignments included inactive areas such as water and agriculture (ecoregion 1) as well as inactive areas outside of our study area.

Initial species selected from FIA data included 10 species which represent greater than 90% of the trees censused along with four additional species known to be key species in succession dynamics in the pinelands (Scheller et al. 2011). Life history attributes were derived from expert knowledge of US Forest Service and research personnel working in the Pinelands, see Table 2 (Scheller et al. 2008; Scheller et al. 2011).

Biomass Succession Extension

The biomass succession extension of LANDIS-II requires numerous physiological inputs in order to assess biomass change across the landscape at long term time scales. All species parameters (leaf longevity, woody decay rate, age related mortality, and the growth curve parameter (used to determine rate of ANPP increases) are from Scheller et al. (2008) and Scheller et al. (2011). For each ecoregion the species establishment probability (SEP), maximum aboveground net primary productivity (ANPP g biomass/ m^2/ year) maximum biomass (MaxB g biomass/ m^2), and actual evapotranspiration (AET mm) are required in order to estimate biomass for all species across the landscape at each timestep. We computed ANPP and SEP for our study area via the program PnET-II for LANDIS-II (Xu et al. 2009) which was specifically designed to output these parameters in the form of LANDIS-II model input. ANPP is a direct output of the base PnET-II model, but SEP is calculated using the product of light and water availability, vapor pressure deficit (also base PnET-II outputs), and a growing degree day parameter specific to PnET-II for LANDIS-II (see below, Xu et al. 2009). MaxB and AET were calculated from field data obtained from USFS personnel working on carbon allocation studies in the Pinelands (Steve Van Tuyl, personal communication).

PnET-II for LANDIS-II requires numerous vegetation and site specific inputs for a successful output, most of which were derived from PnET-BGC documentation and examples (Aber et al. 1997; Aber et al. 1995; Gbondo-Tugbawa et al. 2001). For those parameters in PnET-II for LANDIS-II that were unavailable from any previous PnET source (Aber et al. 1997; Aber et al. 1995; Gbondo-Tugbawa et al. 2001; Scheller et al. 2008; Scheller et al. 2011), namely site specific maximum and minimum growing degree days for the dominant species within each functional group, we accessed LINKAGES model documentation and obtained these parameters for the dominant species of each functional group (Botkin et al. 1972; Pastor and Post 1985). Example physiological data were used for each of our fourteen native species by assigning them to one of the following functional groups: northern hardwood, southern hardwood, spruce fir or pine. Functional group physiology parameters provided as example data from PnET-II has been successfully validated in numerous studies using PnET in northeastern forests (Aber and Federer 1992; Aber et al. 1997; Aber et al. 1995). We assigned our species to the above functional groups following the Scheller et al. (2008) study which also used PnET to obtain ANPP for LANDIS. We assigned species not included in the 2008 study to one of the above functional groups based on range and species type using the US Forest Service Silvics manual (Burns and Honkala 1990). Other requirements for PnET-

II for LANDIS-II include average monthly PAR and WHC for each ecoregion. PAR for our study was obtained from the National Renewable Energy Laboratory (http://www.nrel.gov/gis/data_solar.html, accessed Sept 16, 2011) at 40km resolution (George and Maxwell 1999). Solar information was intersected with ecoregion areas and summarized on a per month basis via ArcGIS10 (ESRI 2011). Conversion of PAR from kW m^{-2} to µmol $m^{-2}s^{-1}$ needed for PnET-II for LANDIS-II input (dividing by .0046) was based on Zheng et al. (2008). WHC for ecoregions was based on spatial data obtained from Pan et al (2009) via Scheller (2011) and extracted for our ecoregions and study area. Climate data (maximum and minimum monthly temperature and average monthly precipitation) was obtained from a 30-year dataset at Atlantic City, NJ, used previously for our region (Scheller et al. 2011).

With PnET-II for LANDIS-II outputs we can create the input table for the biomass succession extension which include SEP and MaxANPP for each species and specific to each ecoregion.

Dynamic Fire and Biomass Fuel Extensions

The dynamic fire extension of the LANDIS-II model requires several spatial and parameter inputs. A fire ecoregion map delineating areas that had burned previously was based on our large (> 40.469 ha or 100 acres) wildfire database (see Chapter 1) for the years 1924-2007. To create the fire region map we overlaid all large fires to create an area in which active large wildfires were known to occur over the last 80 years. We excluded areas of prescribed fire from our fire region since these areas are generally maintained by specific rotation periods and are not affected by changes in wildfire regimes. These areas could be superimposed upon model results to override final fire regime and forest cover results. Large wildfire data from the last 20 years of available data (1987-2007) was used to calibrate our base fire regime to best emulate modern wildland fire fighting practices. We used only the large fire database (see Chapter 1 for description of the fire history geodatabase) due to our previous findings indicating that ignition densities (and small fires) are not linked to landscape level forest composition change in our area. Fire regime calibration requires numerous repeated model runs with different location and scale parameters (the lognormal equivalent of the mean and standard deviation of our fire size distribution) in order to arrive at average and standard deviations similar to fire sizes of the 20 year modern fire regime. The baseline fire regime calibration was determined to be successful when alternate location and scale values did not improve fire regime outcomes. An additional fire ecoregion was created for scenarios based on the ecological wildland urban interface (EWUI) buffer area around altered land (see Chapter 1). The EWUI distance values of 480m and 240m were reduced slightly to 400m in Barnegat and 200m in the Mullica River watersheds due to the restriction of our 100m cell size from LANDIS-II inputs of previous studies. However, with a reduction in the EWUI buffer areas we are providing a conservative estimate of the EWUI influence. Since 45-60% of all of the land in the EWUI from Chapter 1 did not experience large fire in the last 80 years, and fire frequency across the study area has steadily decreased in the last century, all fire was eliminated in the EWUI fire ecoregion to emulate a continued reduction of large fire frequencies in the EWUI. Thus changes in fire regimes for our scenarios were constrained by the area in which fires had burned in the fire history (historic fire region), the EWUI area (EWUI fire region), altered land, fire weather, and the fuels available. Fire weather inputs emulated the Scheller (2011) study in order to promote comparisons between different LANDIS-II scenarios. Fire weather parameters based on codes developed to indicate short and long term weather variability and its effect on fuels included the fine fuel moisture code (FFMC –short term) and the build-up index (BUI –long term, Van Wagner and Canadian Forestry 1987). The model randomly selects from seasonal records of wind speed and direction associated with ecoregions to model fire spread (Sturtevant et al. 2009). In our case, the fire weather table was applicable to all ecoregions due to a lack of variability in weather across our region.

Fuel parameters were taken directly from Scheller et al (2011) and fuel types were based on Scott and Burgan (2005) fuel type classifications. Extensive details on methods for fuel type designations can be found in Scheller et al (2011). Fuels types included hardwood wetland, mixed wetland, conifer wetland, oak upland, mixed upland, pine upland, pine plains, grass, and young upland scrub.

LANDIS-II Scenarios

Altered land vs. Build-out

Although LANDIS-II has been employed in the Pinelands previously (Scheller et al. 2008; Scheller et al. 2011), the goal of this study was not to emulate historic or future fire regimes, but to elucidate how constraints on the amounts of forest and fuels available for burning would affect the fire regime. Scenarios of LULC change and the EWUI fire regimes related to increased altered land are unique to this study. The baseline scenario of current altered land in 2002 (NJDEP 2006) was created using Extract by Mask in ArcGIS10 (ESRI 2011) on the ecoregions, initial species, and fire ecoregion maps constraining the area for burning and succession available for modeling. Build-out scenarios for the Barnegat and Mullica watersheds (Lathrop et al. 2003; Lathrop and Conway 2001) were used as future alternatives to 2002 LULC (NJDEP 2006) and were created by masking the built-out area predictions further constraining the area for burning and succession. Build-out scenarios from Lathrop et al. (2003; 2001) were updated with the latest open space (fee only) areas from compilations of open-space data from several sources including non-profit and state open space maps (NJCF Open Space Data 2010; NJDEP 2003, 2004; NJDEP 2008a; NJDEP 2008b; NJFARMS GIS Data 2009; USFWS 2009).

Modern fire regime versus added EWUI no-fire zone

EWUI fire buffer areas were based on the distance from altered land specified as having little to no fire in each watershed over the last 80 years (see Chapter 1). EWUI areas were created using the Expand function of the Spatial Analyst extension in ArcGIS10 based on altered land and build-out scenarios separately (ESRI 2011). The EWUI in each watershed was created separately due to differing buffer areas for each and rejoined via Mosaic to New Raster in Spatial Analyst using ArcGIS10 (ESRI 2011). We then reclassified the EWUI (using Reclassify in Spatial Analyst) to a high value integer as preparation for combining the EWUI fire region with the base altered land fire regions. We combined the base fire region map with the EWUI fire region map by obtaining the maximum of each of the fire regions using Cell Statistics in Spatial Analyst in ArcGIS10 (ESRI 2011). All extents and cells were carefully checked for alignment before and after each processing step in order to ensure that map inputs for LANDIS-II hadn’t undergone extraneous spatial shifts at the cell or extent level. All four models (see below) were run for 100 years at 10 year time steps.

Model combinations

We compared current altered land without the EWUI and current altered land with the EWUI as well as build-out without EWUI and build-out with EWUI to understand the range of possibilities for forest disturbance and competition in the Pinelands given different levels of human altered land. Results of all scenarios were assessed for species composition with the focus on coniferous versus deciduous dominance and total biomass via percent total area comparisons between scenarios. Output classes were determined by combining maximum biomass associated with each cohort of each species per cell. Classes that contain the maximum biomass for the cell are assumed to be the dominant species in the cell and the cell is assigned to that class. Classes were: ATWHCED (*Chamaecyperus thyoides*), PINE (*Pinus rigida* and *Pinus echinata*), PINE PLAINS (*Pinus rigida, Pinus echinata, Quercus marilandica* and *Quercus ilicifolia*), LOWDEC (*Acer rubrum, Nyssa sylvatica, Liquidambar styracilfula*), and UPDEC (*Quercus coccinea, Quercus falcata, Quercus prinus, Quercus alba, Quercus velutina,* and *Sassafrass albina*). For mapping purposes, the deciduous classes were combined into one DECID class. The PINE PLAINS class was a combination of pine and oak species common to the unique dwarf pine plains region but these species are also found in close association outside of the pine plains ecoregion. Most areas that contain high levels of PINE PLAINS species biomass are dominated by *Pinus rigida* with *Q. ilicifolia* and *Q.marilandica* found in the understory..

Results

Fire module calibration

After approximately 20 trial runs of the Dynamic Fire and Fuels extension, calibration was tuned to within 14% of the average fire size for the last 20 years of our fire history (modern), to within 9% of the standard deviation of modern fire sizes and 4% for modern average yearly ignitions (Table 3). All calibrations were based on the altered land (baseline) scenario. All 100 year model scenarios were run with the parameters obtained from this calibration. Fire frequency across the landscape was similar to historic frequencies with a maximum fire frequency of two times burned. Figure 1 compares the modern 20 year fire frequency (1987-2007) to 20 years of modeled fire. Note that the calibrated fire regime depicted in Figure 1, is only one rendition of the stochastic fire process in the Dynamic Fire and Fuels extension for the altered land scenario.

Fire regime comparisons

Results of our four scenarios: altered land, altered land with EWUI, build-out, and build-out with EWUI show different average fire size across the 100 years modeled (Table 4). Build-out scenarios decreased the average fire size and standard deviations (by approximately 35%, from 1061.51 ha to 699.19 ha for mean fire size over 100 years), but increased the average number of ignitions per year (from 1.59 to 1.94). This means that with increased fragmentation the model generally forecasts more, smaller fires. However, ignitions decrease with the build-out EWUI scenario back to baseline altered land levels so that with a large enough increase in the no-fire EWUI zone, available ignition sites decline. Adding the fire-free EWUI zone to the altered land scenario decreased average fire size by half and adding EWUI to the build-out scenario decreased fire 60%, although fire input parameters remained the same in the main fire region for all scenarios.

Coniferous versus Deciduous composition

The baseline altered land scenario shows forest composition at time zero (the initial communities map) to be a balance of deciduous and coniferous cover with pine plains vegetation classes in atypical pineland areas as a result of randomly assigning one associated FIA plot data to entire forest type polygons (Figure 2). At time 100 we see a drastic decrease in coniferous cover and a shift in pine plains cover towards the pine plains ecoregion with deciduous cover dominating the landscape. The build-out scenario is similar to the altered land scenario at 100 years in that most of the coniferous and pine plains cover is constrained to small extant populations. Constraining the area available for large fires via the EWUI for altered and build-out scenarios did not lead to a dramatic change in coniferous, pine plains, or deciduous cover at 100 years compared to non-EWUI scenarios. However, there is a clear shift from extant islands of pine in the fringe areas of the pinelands to a concentration of the patches in the core of the natural area (Figure 3). Both altered EWUI and build-out EWUI scenarios are almost entirely deciduous cover in the northern half of Barnegat bay where the EWUI is more extensive due to the complex configurations of altered land in this area. The southern area of the Mullica River watershed is impacted by the EWUI as well, with oak cover overtaking pine cover in this fragmented landscape compared to the non-EWUI build-out scenario.

Biomass comparisons

All scenarios depict a gain in percent of total biomass for deciduous species and a loss for coniferous species (Figure 4). Decreases are similar across all scenarios with similar rates of change. The percent biomass comparisons show a clear switch in biomass dominance from coniferous species to deciduous species by year 50. All coniferous and deciduous categories increased in total biomass for each timestep until approximately 70 years into model simulations, at which point pine species asymptote and decrease slightly from year 90 to 100, whereas oak biomass continues to increase until year 100 (data not shown, but see Chapter 4). Cohort age generally increased for most species including *Pinus rigida* (Figure 5).

Discussion

Fire modeling

We calibrated our fire module which was based on a modern fire regime between 1987-2007 to emulate the modern fire regime. We ran the baseline model for 20 years to match the time period from which we accessed our modern fire regime statistics, and in doing so, we were able to adjust the lognormal location and scale parameters to best match the reality of the fire regime on the ground.

When compared to our 20 year modern fire frequency (Figure 1), we see that the modeled 20 year fire frequency for the base altered land scenario is similar to historic levels, although the modeled fire regime covers more of the available area for burning. However, these two maps are not directly comparable in that the actual fire history had a much larger area for burning before the modern configuration of altered land became reality. Five of 30 actual modern large fires extended outside of our watershed boundary and all but the most recent fires burned through land that has since been altered. The Dynamic Fire and Fuels module was limited to starting and spreading within our modeled fire region (restricted by the watershed boundaries and current altered land); therefore all fires are started and spread only within the study area.

Fire regimes between different scenarios in the model were reflective of the amount of land available for burning. When larger swathes of land were available, larger fires were more common. With increasing fragmentation and a fire-free buffer near altered land (EWUI), ignitions increased only slightly for altered EWUI and build-out scenarios but fire size decreased from our average baseline of 1430 ha to our buildout with EWUI scenario of 269 ha. Standard deviations of fire sizes also decreased with increasing fragmentation and EWUI effects.

Coniferous versus deciduous cover

A primary objective of this project was to understand the range of large scale forest composition change we might expect with the differences in altered land and the incorporation of the EWUI concept; where wildfire is reduced to zero at certain distances from altered land. The rate of change of the dominant cover from coniferous to deciduous across the entire study is relatively rapid (approximately 50 years) compared to Chapter 1 coniferous to deciduous transitions (e.g. within 200 years at 143 ha/yr from 1986-2002 within the Barnegat EWUI). However, using the calibrated fire regime based on modern fire suppression practices likely contributed to the rapid modeled changes which have not been seen previously. Changes in forest composition from pine to oak dominance were also found in pre-colonial versus current day forecasts in a previous LANDIS-II modeling efforts in the Pinelands of NJ (Scheller et al. 2008). The Scheller et al. (2008) study simulated current fire management compared to pre-colonial estimations of the fire regime and depicted a landscape changing from a pine-dominated to oak-dominated system. Additionally, pre-colonial estimations of the fire regime were applied to the current landscape in an attempt to model the effect of fragmentation.

However, the changes from pine to oak in the current landscape with pre-colonial fire could not be attributed to fragmentation alone. Our study uses modern fire regimes on a modern landscape and limits fire regions based on altered land, build-out forecasts and EWUI areas with noticeable differences in fire regimes due to increased fragmentation. Additionally, Scheller et al. (2008) explicitly modeled a doubling of the prescribed fire regime currently in place, a threshold which pushed the system back to an estimated pre-colonial pine dominated system. Although we did not model prescribed burning and limited our study to wildfire modeling, the amount of wildfire modeled under current and future scenarios of altered land and EWUI effects was not sufficient for maintenance of pine dominated forest. While modeling prescribed burning in addition to wildfire was outside the scope of this research because we were interested in the effects of altered land on wildfire regimes, doing so must be a research priority as the practice likely represents the only way to manage for pineland vegetation.

The islands of pine dominant forest that remain in all of the model scenarios merit further investigation. The physical and biological limits in the model such as the water holding capacity in each ecoregion, growing degree days, light availability and maximum biomass of each species in each ecoregion help determine the ability of pines or oaks to dominate in these areas. The increase in deciduous biomass from aging trees (see below) decreased the dominance of coniferous cover seen in our 100 year outputs. Although coniferous trees may still be present in these deciduous dominated areas, deciduous biomass is a much higher proportion of total biomass in each cell at 100 years and is the dominant forest canopy species.

Biomass change

Understanding the carbon sequestration implications of total biomass shifts from pine to oak species is complex. As noted, Clark et al. (2009a) found that net ecosystem exchange between pine versus oak dominated forests in the Pinelands were similar near eddy flux towers when averaged across an entire year, except when oak species were defoliated by gypsy moths. Additionally, Miao et al. (2011) found that the Wx-BGC process model accurately predicted similar yearly average carbon flux at three eddy covariance flux towers in pine and oak dominated forest types the Pinelands except when oaks were affected by gypsy moth defoliation. For pine to oak transitions, we may find that defoliation from pests comprise a large loss of net carbon uptake which we did not model as a disturbance in this study. Additionally, we did not model climate change (but see Chapter 4) which could change fire frequency and differentially affect the photosynthetic abilities of different species encountering higher temperatures and carbon dioxide concentrations in the atmosphere. Overall, the general increase in biomass for all cover types over the 100 years modeled is similar to a previous study (Scheller et al. 2011) in our area which concluded that the Pinelands may still be recovering from intense disturbances and deforestation in the early part of the 18th century. However, most of our study area has burned in the last 80 years and initial community age cohorts were based on FIA data from 2005 to 2009, therefore the reduction in fire with our modeled fire frequency (based on the modern fire regime) is most likely driving the overall increase in age and biomass in our results. Coniferous species in our study did see a slight decline in biomass at year 90 indicating that the lack of fire may have lead to a decrease in the establishment ability of pine species with increased canopy biomass.

One significant caveat in our ability to model biomass and forest cover change via LANDIS-II is the unique ability of the dominant pine (*Pinus rigida*) to resprout epicormically from the bole and branches of the tree after less severe wildfires. Currently in the Biomass Succession extension for LANDIS-II, fire removes younger cohorts at lower severities and removes older cohorts with increasing fire severities. If the species resprouts, the model immediately re-establishes that species at a cohort of age zero. In reality, *Pinus rigida* would not be starting at age zero and would retain much of its woody biomass at certain fire severities, losing mainly its needle leaves and smaller branches. Consequently, the biomass projections for *Pinus rigida* and in a related manner, the other succession functions dependent on age cohort calculations (such as shade tolerance), are most likely significantly under-represented in the model and could be highly under-estimating the persistence of pine cover. However, recently, epicormic repsrouting has become available in the Century Succession extension of LANDIS-II. Future modeling efforts should utilize this recent modeling development for the Pinelands ecosystem.

Management Implications

In Chapter 1 we found that altered land results in decreased fire frequency and a transition from pine to oak forest composition within the EWUI between 1986 and 2002 based on the entire fire history available. In our modeling scenarios, we based the fire regime on the years 1987 to 2007 and our initial conditions for forest cover and cohorts were within the range seen in the 2005-2009 FIA survey. The large shifts in pine to oak cover even without the implementation of the EWUI no-fire zone may be an indication that the change in forest composition seen in the EWUI in Chapter 1 could rapidly expand outside of this area given the modern fire regime. The modern fire regime altered and buildout scenarios represent scenarios whereby 45-60% of the areas near altered land had not burned in the last 80 years, but in which some burning has occurred. Inclusion of fire in areas near altered land based on recent fire history could be useful in forecasting the amount of fire needed to maintain pinelands near altered land. The no-fire EWUI scenario represents a future in which suppression efforts are highly effective and trends showing decreases in fire frequency and size across the 80 year fire history continue. The scenarios that include the no-fire EWUI may be more indicative of future conditions where fire is not allowed to burn near life or property. The altered land with EWUI and the build-out with EWUI scenarios show even lower average fire sizes with more transition to oak dominated forest cover. The Pinelands Comprehensive Management Plan specifies that, "the continued integrity of the Pinelands vegetation is essential to the preservation and maintenance of the essential character of the Pinelands;" therefore active maintenance and ecological restoration, through prescribed fire or other means must be implemented for a large part of Pinelands vegetation to persist.

Comparisons to relevant studies

Human caused disturbance in the form of fire on forests has been significant (Bradshaw and Hannon 1992; Delcourt and Delcourt 1998; Syphard et al. 2007), with long term, large scale changes to forest composition. Modeling of changes in fire regimes has successfully forecast changes in forest composition, but has focused on general changes in regimes rather than varying human impacts to fire regimes across the landscape (Bergeron et al. 1998; Scheller et al. 2005). Other studies focus on the expansion of the human influence on natural areas via housing density forecasts (Hammer et al. 2007;

Nowak et al. 2005) in the wildland-urban interface. Studies of urban influences on forests have generally focused on urban to rural gradient effects on forest patches (Zipperer 2002), or have focused on urban ecosystems as a unit (Pickett et al. 2008) rather the effect of human influence on adjacent, contiguous forested areas. Our research fills a unique niche by modeling both the expanding human influence via buildout models and the ecosystem response to reduction of fire from suppression in areas of known human influence (the EWUI fire region). Modeling the variable gradient effect of the EWUI and its exacerbating effects on fire suppression near altered land using empirical data of fire frequency is a new approach to understanding the effects of human influence on the ecology of natural lands. Using this novel approach we were able to assess the potential for human caused disturbance to shift forest composition and change ecosystem function at the landscape level.

Conclusions

Natural systems do not exist in a vacuum, separate from human influence. Critical understanding and effective management of natural systems will require an equal or greater perception of the human dimension of influence. Our major findings showed a decrease in average fire size due to increased fragmentation and a further decrease when fire suppression on adjacent natural land was incorporated (via the no-fire EWUI area). Additionally, we found that the shift from a pine dominated system to an oak dominated system in the absence of fire was accelerated near altered land in EWUI scenarios. The lessons learned in this case study of complex disturbance dynamics are important indicators of the coupled human-natural system. Our efforts to conserve Pinelands composition will be unsuccessful if approached in a manner that excludes the history of human influence on fire frequency. Additionally, if increasing fragmentation is not addressed, it will be difficult to use ecological disturbance processes such as fire as part of our restoration management approach.

The ecological effects of surrounding development on natural areas are ever-increasing and the gradient of human influence dynamic is becoming more evident (Kauffman 2004). Using a spatial model to understand the extent to which development influences ecosystem function provides an essential decision support system for social and environmental policy and can be updated to provide tools in an adaptive management approach. This type of long term spatial disturbance and succession modeling facilitates a greater understanding of the linkages between the forces of human-caused change and the ecosystem dynamics associated with this change and leads to a deeper insight into how ecosystem services may be affected. For example, if forest composition in the Pinelands changes to oak dominated forest, will endemic and rare Pinelands plants and animals be lost? Additionally, will soil composition change with increased broadleaf deposition and decomposition and will that affect filtration and water quality? Will decreased fire and increased fuels create a catastrophic fire danger in which fires become more and more difficult to actively suppress? Although we concentrated on general classes of forest cover and not other species associated with these canopy differences, canopy cover generally influences shade levels and nutrients available for understory species, inherently producing entirely different suites of species in the understory (Keith et al. 2010; Nowacki and Abrams 2008).

It seems likely, given our model results, that this highly human influenced ecosystem and its future will be determined by our management decisions concerning altered land and fire. It is our hope that this study will assist in understanding where management efforts should be concentrated across the landscape and facilitate setting goals for thresholds needed to maintain Pinelands ecological integrity. For example, in areas that transition to oak cover in all scenarios, prescribed fire and/or mechanical treatments that mimic fire may be necessary to maintain pine cover and thresholds of fire frequency, area burned, average fire sizes, or understory treatments determined. Given that this area was originally protected for its “essential character and ecological values” (Pinelands Commission 1981, revised 2010), it will be necessary to revise current land use and fire management plans in order to address the potential major changes in future forest cover. These results will not only assist local managers but may be applicable to areas such as the Southeastern US due to the large amount of wildland/urban interface in fire prone pinelands of that region (Hammer et al. 2009).

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Table 1. Water holding capacity for each ecoregion. Values used as inputs for PnET-II for LANDIS-II in calculating ANPP and SEP for each functional type in each ecoregion.

Description	Ecoregion	WHC (cm)
UpLow	2	6.965
UpMed	3	7.361
UpHigh	4	7.680
WetLow	5	7.469
WetMed	6	7.933
WetHigh	7	8.437
Plains	8	7.236

Table 2. Initial communities species used along with life history attributes needed for the LANDIS-II model.

Species	Longevity (yrs)	Age at maturity (yrs)	Shade tolerance	Fire tolerance	Effective seeding distance (m)	Max seeding distance (m)	Probability of resprout	Min resprout age (yrs)	Max resprout age (yrs)	Post-fire regeneration
Acer rubrum	150	10	4	1	100	1000	0.5	10	140	none
Chamaecyparis thyoides	400	12	3	3	183	1000	0.5	5	100	resprout
Nyssa sylvatica	200	15	4	2	30	1000	0.75	0	100	none
Pinus echinata	200	20	1	3	60	500	0.75	5	25	resprout
Pinus rigida	200	5	1	3	60	250	0.75	5	60	resprout
Quercus alba	300	40	3	3	30	3000	0.5	5	40	resprout
Quercus coccinea	120	20	2	1	30	500	0.5	5	75	resprout
Quercus falcata	150	25	3	2	30	500	0.75	5	25	resprout
Quercus prinus	200	20	3	3	30	500	0.75	5	60	resprout
Quercus velutina	250	20	3	2	30	3000	0.4	5	25	resprout
Liquidambar styraciflua	350	25	2	2	60	180	0.75	5	50	resprout
Sassafras albidum	150	10	2	2	30	3000	0.75	5	115	resprout
Quercus ilicifolia	50	5	1	1	30	500	0.75	5	50	resprout
Quercus marilandica	150	5	1	1	30	500	0.75	5	150	resprout

Shade and fire tolerance are on at a scale from 1 (least shade tolerant/least fire tolerant) to 5 (most tolerant). Data derived from Scheller et al (2008, 2011).

Table 3. Mean, standard deviation and the mean yearly ignition count for 1987-2007 (modern fire regime) as used for calibration of the Dynamic Fire and Fuels module of LANDIS-II. Also shown are the final calibration values used in fire modeling for all scenarios as well as the 20 year fire calibration output and percent difference from the actual modern fire regime.

Modern fire regime			Final Calibration		
mean (ha)	stddev (ha)	mean igtn/ yr	mu	sigma	mean igtn/ yr
1065.170	1958.719	1.500	9.000	0.880	6.800
20 year modeled output			Percent difference		
mean (ha)	stddev (ha)	mean igtn/ yr	mean	stddev	mean igtn/ yr
1226.414	1797.160	1.450	14.073	-8.603	-3.390

Table 4. All scenarios used the final 20 year modeled fire regime calibration. Increased fragmentation (build-out) no fire for buffers around altered land (EWUI) reduced average fire sizes and standard deviations over 100 years of fire and succession.

	Final Calibration			100 year output		
Model scenario	mu	sigma	mean igtn / yr	mean (ha)	stddev (ha)	mean igtn / yr
Altered land	9.000	0.880	6.800	1061.509	1430.363	1.590
Altered with EWUI	9.000	0.880	6.800	523.506	1021.078	1.740
Buildout	9.000	0.880	6.800	699.191	936.319	1.940
Buildout with EWUI	9.000	0.880	6.800	268.766	510.661	1.580

List of Figures

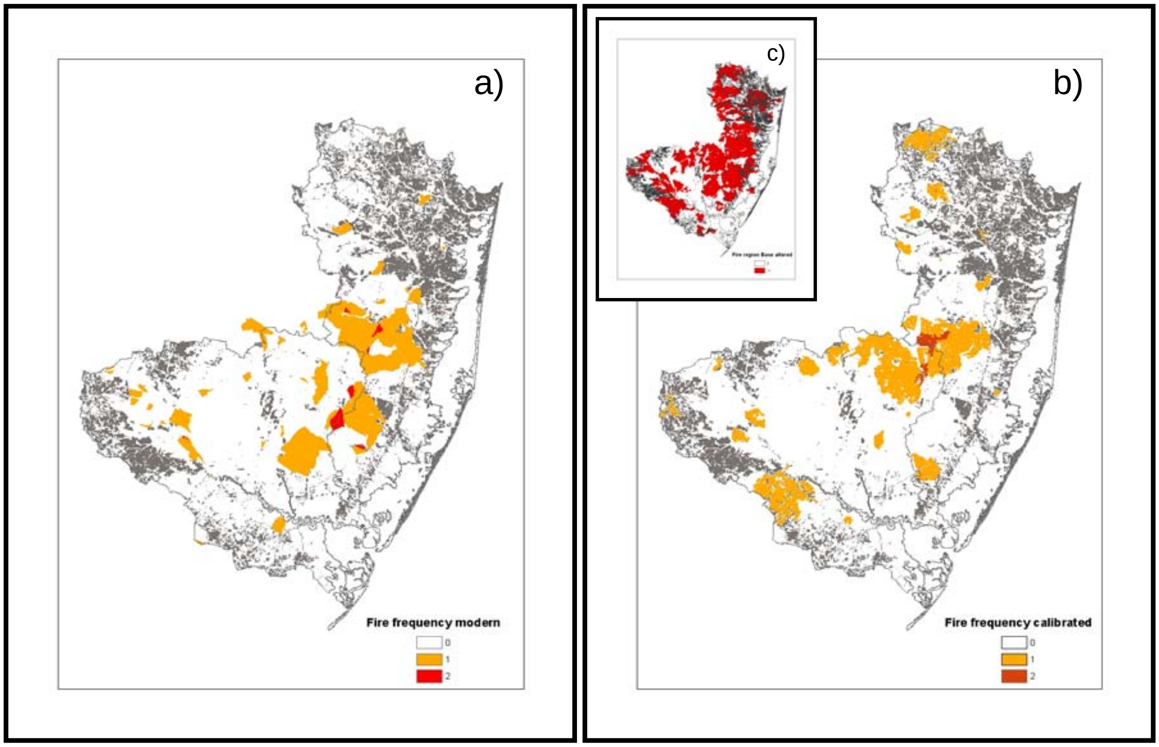
a)
Fire frequency modern
0
1
2
c)
b)
Fire frequency calibrated
0
1
2

Figure 1.

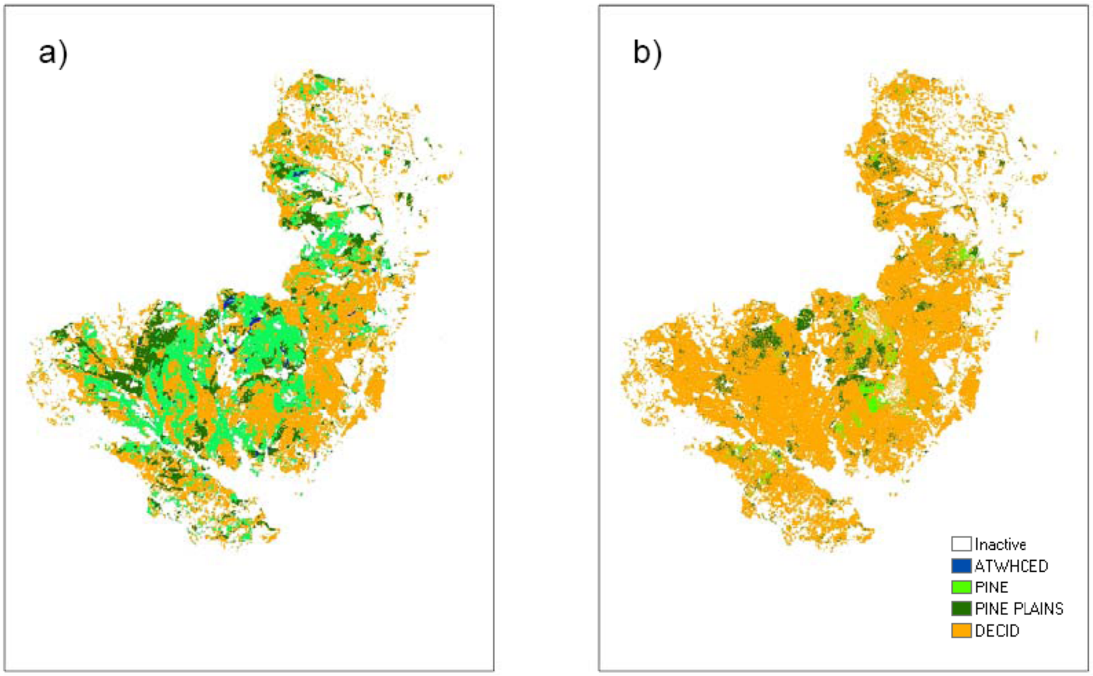

Figure 2.

Figure 3.

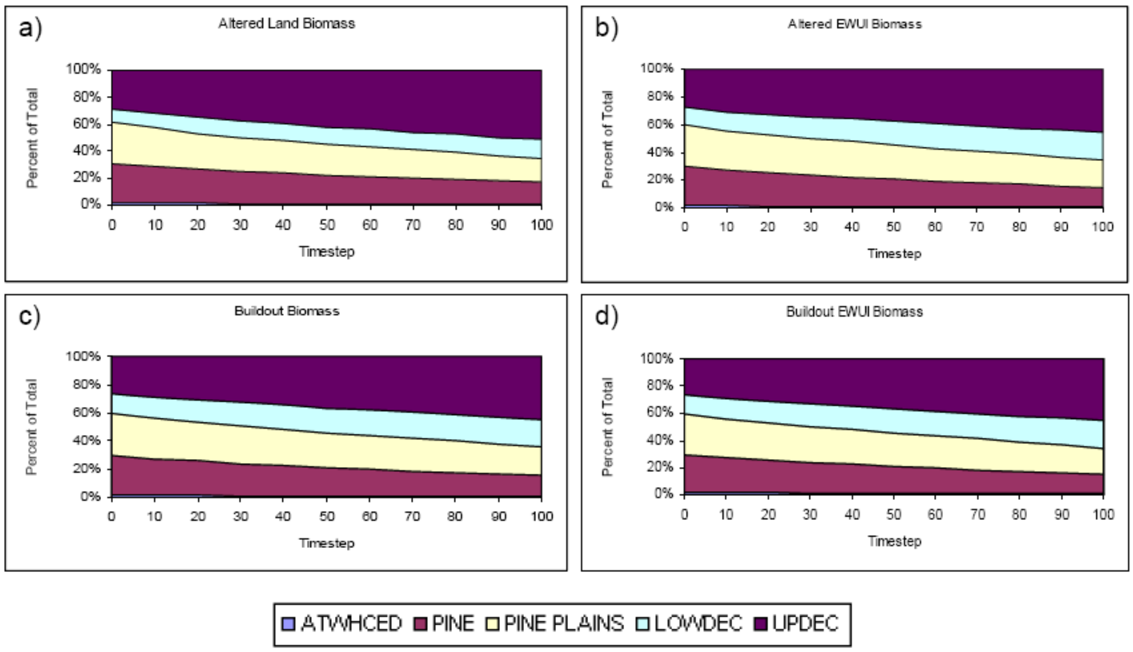
a)
Altered Land Biomass
b)
Altered EWUI Biomass
c)
Buildout Biomass
d)
Buildout EWUI Biomass
Percent of Total
100%
80%
60%
40%
20%
0%
0 10 20 30 40 50 60 70 80 90 100
Timestep
ATWHCED
PINE
PINE PLAINS
LOWDEC
UPDEC

Figure 4.

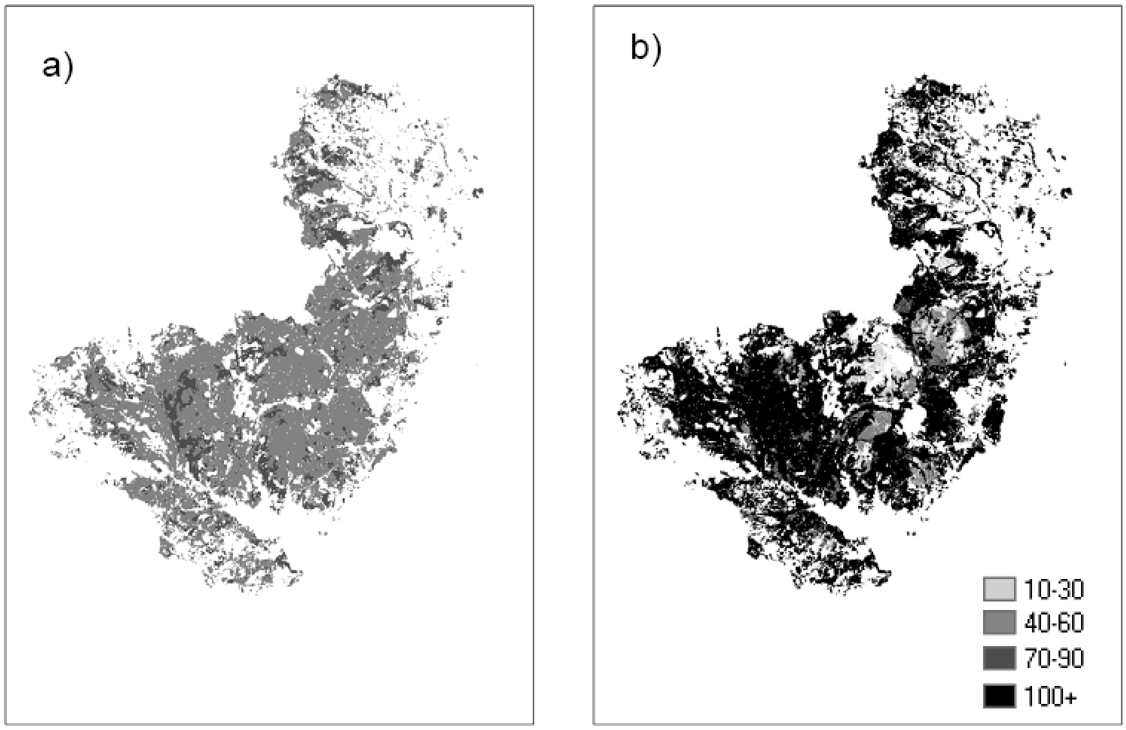

Figure 5.

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CHAPTER 4

Islands of pine: Future climate scenarios in the NJ Pinelands using the LANDIS-II forest landscape disturbance and succession model

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Abstract

Assessing forest resilience to disturbances including climate change is an important aspect of adaptive management. Climate change impacts on fire and forest composition in the Pinelands of New Jersey have not been assessed to date and the prospect of major shifts in forest composition present challenges to the mission of the Pinelands National Reserve to preserve the ecological integrity of the Pinelands. The balance of pine versus oak cover may have long term effects on accumulated biomass and other ecosystem services. Ecological interactions between reduced fire due to nearby altered land, available forested area for burning, and climate change effects on fire will determine fire size and frequency. Factors such as increases in aboveground net primary productivity and changes in species establishment probabilities in a warming climate are also key determinants of pine versus oak forest composition. We used climate change forecasts in combination with a landscape disturbance and succession model (LANDIS-II) to investigate the interactions between climate and altered land forecasts, fire regimes, and forest composition. Our modeling results suggest that with increasing temperature and CO_2 the average fire size in the Pinelands of New Jersey will mimic base altered land scenarios and forest composition will change from pine to oak cover. Results can be used for incorporating climate change into adaptive management plans for Pinelands preservation efforts and as an example of a protected ecosystem highly dependent on human management decisions in an urbanized context.

Introduction

The most recent IPCC summary report (IPCC 2007) concludes that increasing carbon dioxide (CO_2) levels increased global mean temperatures by .74 ºC over the last 100 years and will continue to raise temperatures into the future unless mitigating actions are implemented. An adaptive management approach and understanding resilience thresholds are ways to address ecosystem challenges induced by a changing climate for forested ecosystems (Chmura et al. 2011; Millar et al. 2007). Adaptive management incorporates both social and scientific aspects by approaching management decisions using current science and models at the appropriate scales and accounting for various stakeholders interests (Allan and Stankey 2009). Strategies for adaptive management include managing for the capacity of an ecosystem to return to desired states after disturbance (i.e. resilience, Millar et al. 2007). Forest landscape models are an important tool for assessing potential ecosystem resilience to disturbance, including climate change (Dale et al. 2001; Mladenoff 2004; Pearson and Dawson 2003). In forest landscape modeling, aspects such as above ground net primary productivity (ANPP) (Boisvenue and Running 2006) and available habitat for species establishment (Iverson et al. 2008; McKenney et al. 2007) can determine how successful a species is in a changed environment and the rate at which it can adapt. Additionally, changes in precipitation, humidity and temperature contribute to short term fine fuel moisture and long term drought effects (such as the build-up index) and alter spatial and temporal signatures of wildfire disturbance on the landscape (Stocks et al. 1998). In Chapter 3 we found that a decrease in the fire regime due to an increase in the amount of developed land could drastically change modeled Pinelands forest composition and biomass over time, increasingly favoring oak dominated forest over pine. We found in Chapter 1 that altered land begets a lower fire frequency through fire suppression in adjacent natural areas and that this suppression leads to transitions from pine to oak in the ecological wildland urban interface (EWUI). The species that can most effectively handle increased CO_2 and temperature along with the effects of fragmentation and shifts in fire disturbance regimes will have a competitive advantage in the Pinelands of New Jersey. In order to better understand the range of possible future scenarios for our study area, we have incorporated climate change as an additional human caused disturbance in our ecosystem modeling to investigate both bottom-up (fire and altered land) and top-down (climate) disturbance effects on forest composition.

Landscape level changes in forest composition may also have climate feedback implications, in that the timing and intensity of carbon sequestration in pine vs. oak cover could be highly variable (Pan et al. 2009). Thus, if more carbon is released due to increased decomposition of litter (Shi et al. 2009), or if more carbon is sequestered due to differing plant physiology (Miller-Rushing et al. 2009; Schäfer 2011), overall climate change effects may be dampened or enhanced . This chapter aims to forecast potential effects of increasing temperature and CO_2 on pine versus oak composition. Additionally, potential changes in biomass and related carbon storage across the landscape will be assessed. Forest composition changes alone, e.g., from pine to oak, do not clearly indicate more or less carbon uptake. A study by Clark et al (2009a) demonstrated no average yearly difference in carbon uptake near flux towers in pine dominated vs. oak dominated forests in the Pinelands. Process model results for NPP and NEE also show little difference in pine versus oak dominated mid-Atlantic forests (Miao et al. 2011; Pan et al. 2009). However, when other factors are included, such as differential effects of increased temperature on stomatal conductance (Griffin et al. 2000; Roberntz and Stockfors 1998), maximum biomass potential (Poudel et al. 2011; Wertin et al. 2011), species establishment probabilities (Mohan et al. 2009; Scheller and Mladenoff 2008), and altered fire regimes (Marlon et al. 2009; Westerling et al. 2006); the potential for these changes to interactively and dramatically change carbon uptake trajectories is significant.

In the LANDIS-II spatial succession and disturbance model, the mechanisms of climate change include dynamic above ground net primary productivity (ANPP) and species establishment probabilities (SEP). These variables change through time according to a specific climate forecast. ANPP and SEP must be computed for each species or functional group in each ecoregion under a climate change scenario via process models such as PnET-II for LANDIS-II (see Chapter 3 for details on PnET-II for LANDIS-II, Xu et al. 2009). Inputs for climate change scenarios with PnET-II for LANDIS-II include temperature and CO_2 forecasts and a binary parameter determining whether increasing CO_2 concentrations affect stomatal conductance for each functional group. Dynamic timestep outputs for each species within each ecoregion are used directly for input into LANDIS-II climate change forecasts.

When parameterizing physiological process models such as PnET-II for LANDIS-II, we must take into account that the effects of increased temperature and CO_2 on stomatal conductance and net assimilation rates are variable depending on leaf structure and water availability (Griffin et al. 2000; Roberntz and Stockfors 1998;

Warren et al. 2011). The balance of CO_2 and available water in leaves determines the rate of photosynthesis. Increases in atmospheric CO_2 allows leaf stomata to close (decreasing stomatal conductance) which decreases transpiration and water loss from leaves (and increases water use efficiency). Precipitation patterns, temperature, soil moisture, leaf to air vapor pressure deficit, and nitrogen limitations are all factors that will determine the ability of trees to take advantage of higher CO_2 and increased water use efficiency (Centritto et al. 2011).

There is some debate over the ability of pines to decrease stomatal conductance and increase the efficiency of photosynthesis with increased CO_2 concentrations (Haworth et al. 2010; Roberntz and Stockfors 1998; Runion et al. 1999), although a meta-analysis of 13 studies on tree species in Europe found no significant difference in stomatal conductance of conifers with increased CO_2 (Medlyn et al. 2001). The sole pine species, *Pinus sylvestris* included in the meta-analysis showed no difference between ambient and elevated CO_2 for both ambient and elevated temperature treatments (Medlyn et al. 2001). Experiments using open air CO_2 increases on *Pinus taeda* found a higher photosynthetic increase in 1st year needle-leaves versus 2nd year leaves (Ellsworth et al. early view). In terms of the resulting biomass from increased photosynthesis, mature Norway spruce stems in Sweden did not increase in diameter with the interacting effects of higher CO_2 and temperatures in field-grown trees (Kostiainen et al. 2009). Oak species universally decreased stomatal conductance, increased water use efficiency, and increased growth with higher CO_2 levels (Li et al. 2007; Lodge et al. 2001; Medlyn et al. 2001).

Another advantage oaks may display over pines is there ability to utilize higher temperatures and longer growing seasons (Botkin et al. 1972; Pastor and Post 1985). Growing degree days for a specific species accumulate based on daily minimum and maximum temperatures needed for growth and are one way to account for the affects of temperature on photosynthesis rates and biomass accumulation. Some forest models use growing degree days to determine temperature thresholds and their affects on photosynthesis rates of specific tree species or functional groups (Botkin et al. 1972). For example, the dominant oak species (*Quercus alba*) may reach the minimum growing degree day thresholds sooner and reach the maximum growing degree day threshold later than the dominant pine (*Pinus rigida*) and can accrue greater amounts of biomass with an extended growing season due to higher and lower temperature tolerances (Botkin et al. 1972; Pastor and Post 1985). Additionally, with increased temperatures, seedling establishment may be negatively affected when optimum photosynthesis temperatures are surpassed. Based on the literature, our current understanding is that our dominant oak species are generally more adept at utilizing higher CO_2 concentrations and temperatures in comparison to our dominant pines. However, this does not preclude the possibility of a southern pine such as loblolly pine performing better under climate change and migrating into the area.

Studies integrating increased CO_2 along with the factors of precipitation, soil moisture, vapor pressure, and temperature effects on pine versus oak trees in the field are lacking. Typically the most common way to address the difficulty of setting up outdoor experiments for all of these factors is to investigate these interactions via physiological process models that are designed to account for all known mechanistic leaf traits in relation to the environment.

In trying to specify the differences between pine versus oak cover, PnET-II (the basis for PnET-II for LANDIS-II) can be used to model the relationships between temperature, elevated CO_2, and water availability and assumes a doubling of water use efficiency with a doubling of CO_2 for species able to take advantage of the doubling of CO_2 (Aber et al. 1995). For PnET-II validation, aboveground net primary productivity for foliage and wood was within 5% of in situ eddy flux tower measurements (Aber et al. 1995). One drawback to PnET-II is that biomass production and litter are decoupled; therefore fertilization and respiration due to increased decomposition may be underestimated in a future climate (Aber et al. 1995).

Climate change forecasts include increased temperatures, but change may also be expressed as a more variable climate with increased likelihood of droughts or extreme precipitation events (Karl et al. 2008). Climate change induced droughts may increase fire frequency through increased fuel flammability (Marlon et al. 2009; Robinson 2009). One of the largest fire seasons on record in the Pinelands (1963) was a consequence of the number of dry months preceding the fire season (Forman and Boerner 1981). Since fire frequency is directly related to pine cover and dry conditions in the Pinelands we expect pine cover to increase with increased wildfire frequency as a result of a warming climate (Little 1979). However, if fire is reduced and more oak cover results via succession under climate change, a positive feedback for continued pine to oak transition could be established, in that oak cover forests typically have lower fuel loadings resulting in further decreases in the probability of fire ignition and spread (Clark et al. 2009b).

Whether a species establishes or experiences shifts in range depends on many factors including soil moisture, soil composition, temperature, precipitation, and wind; and biotic factors such as shade tolerance, seed dispersal, leaf physiology, and disturbance tolerance (Morin et al. 2008; Scheller and Mladenoff 2005; Xu et al. 2009). Given the potential advantage of deciduous species in assimilating increased levels of CO_2 in the face of water limitations as well as higher temperature and shade tolerances, along with greater maximum seed dispersal distances via bird dispersal of acorns (Darley-Hill and Johnson 1981), these species may prove to be better competitors in a changing climate. Alternatively, if wildfire proves to be the dominant process in shaping the Pinelands ecosystem under climate change, conifers may dominate. Our hypothesis for the effects of modeled climate change on the forest composition (pine versus oak cover) in the Pinelands was that more frequent fire due to rising temperatures may increase pine cover, but that net CO_2 assimilation increases in oaks will outpace pine growth, inducing faster rates of oak succession. Due to the reduction in flammability of oak dominated cover, a positive feedback toward oak succession will result in more oak dominated forests throughout the modeled time period.

Methods

All climate data are identical to those used in a separate concurrent LANDIS-II climate change modeling effort for the Pinelands which concerns defoliation due to gypsy moths (Scheller et al. in prep) in order to facilitate comparisons amongst our studies. Scheller et al. (in prep) used the A2 scenario from the IPCC family of scenarios which indicates no future mitigation of climate effects, an increasing population, and little global cooperation (IPCC 2007). This climate forecast will serve as a guide for a potential expectation of disturbance for the Pinelands. These climate data were downscaled from the HadCM3 general circulation model (IPCC 2007), by extracting the data for the location overlying our study area (Scheller et al. in prep). Next, the LARS-WG stochastic weather generator (Tebaldi and Knutti 2007) was employed using the HadCM3 A2 forecast along with modern weather data from our site to generate future weather over the next 100 years. These data were applied to the Pinelands region with no significant differences in the forecast across the Pinelands landscape (i.e. all areas of our study increased similarly in temperature). CO_2 increases over our 100 years of simulation were based on extrapolations from current CO_2 levels to A2 predictions for CO_2 levels for the year 2100 (IPCC 2007).

PnET-II for LANDIS-II

The first step to adding climate change to altered land and EWUI scenarios was to use PnET-II for LANDIS-II to estimate ANPP and SEP via modeling the physiological effects of increasing temperature and CO_2 on specific species or functional groups. As a conservative step, increasing CO_2 was set to have the effect of decreasing stomatal conductance for oaks, but not for pines in PnET-II for LANDIS-II. Additionally, a sensitivity analysis for this parameter was also performed by removing the difference in CO_2 assimilation between pines and oaks and setting both functional groups to have a decreased stomatal conductance and increased water use efficiency with increasing CO_2 concentrations. LANDIS-II utilizes the time stamped and climate adjusted SEP and ANPP parameters from PnET-II for LANDIS-II to affect species potential using a dynamic input table for each species and ecoregion combination. These parameters are accessed at regular time steps during the model run. Fire spread and duration were adjusted for climate change using the precipitation and temperature outputs from the LARS-WG outputs. These weather parameters were used to adjust calculations for the fine fuel moisture code (short term moisture availability) and the build-up index (long term moisture availability) used in the dynamic inputs needed for modeling fire initiation and spread in the Dynamic Fire System module of LANDIS-II (Scheller et al. in prep; Sturtevant et al. 2009; Van Wagner and Canadian Forestry 1987).

LANDIS-II scenarios for climate change

Our scenarios for climate change included comparing the altered land with EWUI and altered land with EWUI + climate change (AEC) trajectories as well as the build-out with EWUI and build-out with EWUI + climate change (BEC) scenarios. We assume that the EWUI will continue to exist in its present state for Barnegat and Mullica watersheds and do not model altered land without the EWUI affect on fire regimes. These scenarios should elucidate how climate change may exacerbate or dampen the rates of change seen in Chapter 3 for fire regimes and forest cover. Analyses include assessing percent forest cover change and rates of change for forest composition and aboveground biomass over 100 years at 10 year time steps.

Results

Our model results largely followed the trends seen in Chapter 3. Following is a description of each parameter affected by climate change.

Fire regime comparisons

With climate change, the average fire size over 100 years decreased (from 523.51 ha to 496.80 ha) approximately 27 ha between the altered with EWUI model run and the AEC model run (Table 1). The standard deviation of fire sizes and the average ignitions per year also decreased slightly. Additionally, the average fire size over 100 years decreased 8 ha (from 268.76 ha to 259.97 ha) between the build-out with EWUI and the BEC scenario. Ignitions per year also decreased slightly with the BEC scenario, as did standard deviations of fire sizes over 100 years. Total area burned over the 100 years modeled decreased with climate change for both the AEC (10,111 ha less than the altered EWUI scenario) and BEC scenarios (4769 ha less than buildout EWUI).

Reduction of pine and pine plains cover was evident at the 100 year timestep for both AEC and BEC scenarios (Figure 1). Loss in pine and pine plains cover from non-climate change to climate change scenarios was 67% for the AEC scenario and almost 70% for the BEC scenario, although non-climate change areas were already small and isolated. A few remnants or islands of pine and pine plains cover remain in our study area for both climate change scenarios and are concentrated in the interior natural areas over the pine plains ecoregion where water holding capacity is lowest and where the dwarf pine plains is currently common. The majority of the landscape for all scenarios is dominated by oak cover.

Climate change severely reduced the proportion of total pine and pine plains biomass from approximately 58% at time zero to 34% at time 100 of total Pinelands biomass (Figure 2). Deciduous biomass increased from 40% of total biomass to 66%. Total biomass varied across the region for all scenarios and the pine plains ecoregion is evident with lower total biomass than the surrounding landscape (Figure 3). Total biomass per year for all ecoregions in non-climate change scenarios generally increased for all functional groups until year 80 when the pine cover types reached maximum and began decreasing slightly in year 90 (Figure 4). Oak biomass increased steadily until year 100 with no indications of an asymptote in our model. Under the altered EWUI/A2 climate change scenarios, pine biomass peaked at year 30 and pine plains at year 70; whereas both deciduous cover types continued to increase in biomass throughout the 100 modeled years. For buildout EWUI/A2 we see a similar pattern of biomass gains and losses. Overall, climate change resulted in a lower biomass after 100 years for pine and pine plains classes than year zero, whereas EWUI only scenarios increased biomass for all classes from year zero to year 100.

Our sensitivity analysis addressed whether the CO_2 affect on stomatal conductance for oaks and not for pines was a major determinant for our forest composition or biomass results. In PnETII for LANDISII, biomass and forest composition were not substantially different when CO_2 fertilization effects were applied to pine species as well. In searching for essential differences between pine and oak species within PnETII for LANDISII regardless of CO_2 effects, the growing degree day tolerances for our dominant pine versus our dominant oak species were found to be critical to understanding differences in the calculation of species establishment probabilities (Table 2). Species establishment probabilities dropped dramatically for pine species to near zero by year 45 but dropped only slightly for oak species from .71 at year zero to .41 at year 100 (data not shown). This difference was the main driver for the inability for pines to grow new cohorts in a changing climate.

Discussion

Our results indicate a slight decrease in both the mean and standard deviation of fire size over 100 years of modeling with climate change. This is in contrast to increased fire regimes in climate change/wildfire modeling studies in Western forests (Spracklen et al. 2009) and analyses of historic increases in fire due to increased summer temperatures in the Western United States (Westerling et al. 2006). One climate change fire modeling study found variable responses to climate change across North America with decreases in fire regimes overall (Flannigan et al. 1998) but this is the first study modeling the effects of climate change on modern fire regimes in the New Jersey Pinelands. In our study, the build-out with EWUI and climate change scenario resulted in the smallest average fire sizes and total area burned suggesting that the effects of increasing fragmentation combined with increasing deciduous dominance could accelerate the reduction in the fire regime for our area under climate change.

Overall, the biomass values for our classes and the range of values absent of climate change are comparable to Scheller et al. (2011) and Miao et al.(2011). The decrease in fire size and standard deviations under climate change combined with the tremendous increase in biomass production by deciduous trees resulted in decreases in pine and pine plains cover for these scenarios. Factors that may have contributed to oak cover success in our climate change model include: decreased stomatal conductance and greater water use efficiency (via the CO_2 stomatal conductance effect in PnET-II for LANDIS-II), higher ANPP and SEPs with a warming climate (i.e. the output of the PnET-II for LANDIS-II model), higher shade tolerance, a greater maximum seed dispersal distance, and a higher maximum biomass for older cohorts. Since our sensitiviy analysis showed that stomatal conductance was not a major factor, temperature tolerances of our dominant pine versus our dominant oak were most likely the driving factor for increased deciduous cover.

Deciduous biomass increased linearly throughout the 100 modeled years whereas pine and pine plains biomass initially increased then decreased in climate change scenarios. Although both deciduous and coniferous trees were aging, deciduous trees increased in biomass at a more rapid rate creating forests dominated by deciduous biomass. Although pines may be present in these forests, they are present in the form of older trees. The drop in pine biomass seen in our climate change scenarios can also be attributed to a lack of fire which reduced the ability of pines to take advantage of forest gaps and regenerate with younger cohorts. Almost all of the cells with *Pinus rigida* contained maximum cohort ages of 40-60 years at time zero. For the AEC scenario, the majority of cells contained maximum cohort ages of 100 years or more (data not shown). Lack of fire inhibited gains in pine species biomass through regeneration and establishment after 50 modeled years and eventually led to a die off of older cohorts reaching maximum age (200) for the species. The lack of epicormic resprout modeling for *Pinus rigida* in our simulations (see Chapter 3) may not be a major factor for final pine biomass projections under climate change conditions if fire is reduced to minimal levels due to increased dominance of oak cover and the amount of altered land.

Our model did not assess the impact of insect defoliation or infestation and the Pinelands have been shown to have substantial differences in NPP based on gypsy moth defoliation of oak species (Clark et al. 2009a; Miao et al. 2011). However, a new threat to the Pinelands (potentially due to climate change shifts in species range), is the southern pine beetle (Shumate et al. 2009). Damage caused to the dominant pine species, *Pinus rigida,* from the southern pine beetle could offset the loss in ANPP to oaks from gypsy moth damage. These issues are being addressed in Scheller et al. (in prep).

Conclusions

Our research provides planners and fire managers with a range of possible outcomes in the Pinelands to inform climate inclusive management plans. As yet, there has been dearth of landscape level information on climate effects in the Pinelands. However, the LANDIS-II model has already been applied successfully in the Pinelands for numerous purposes; and recently, researchers using LANDIS-II in this area have begun planning a combined effort for a 'mega' LANDIS-II run with all possible disturbances and with the best available data from all previous studies in order to refine our forecasts for the Pinelands region. This research helps further that goal by providing an example of the interactions of changes to altered land, the EWUI fire regime, and climate change. Specifically, we found that under our climate change scenario, the fire regime did not increase or decrease substantially given the amount of altered land and the influence of fire suppression near altered land. Additionally, we found that with increased temperatures, temperature tolerances become more important in species establishment probabilities and the ability of pines to establish and produce young cohorts through time.

Ecosystems that persist via fire disturbance throughout the world face increasing challenges under a changing climate. The direct and indirect effects of climate change will be highly variable in terms of maintaining high-value ecosystems and their services Land managers will require the ability to incorporate sophisticated methods for predicting such effects in order to mitigate them and boost ecosystem resilience. This research provides a novel approach by assessing human impacts on the disturbance of adjacent natural areas at the regional scale as well as the global scale via temperature and CO_2 increases. Other human-natural systems would benefit from using this type of scale dependent approach.

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Table 1. All scenarios used the final modern fire regime calibration from Chapter 3 to model fire. Climate change decreased average fire sizes and standard deviations over 100 years of fire modeling for both altered with EWUI and build-out with EWUI scenarios. Additionally total area burned throughout the 100 year scenarios decreased with climate change.

	Final Calibration			100 year output			
Model scenario	mu	sigma	mean igtn / yr	mean (ha)	stddev (ha)	mean igtn / yr	Total area burned (ha)
Altered with EWUI	9.000	0.880	6.800	523.506	1021.078	1.740	91090
Altered with EWUI and A2	9.000	0.880	6.800	496.804	846.567	1.630	80979
Buildout with EWUI	9.000	0.880	6.800	268.766	510.661	1.580	42465
Buildout with EWUI and A2	9.000	0.880	6.800	259.972	311.956	1.450	37696

Table 2. Growing degree day (GDD) parameters used in calculating species establishment probabilities in PnETII for LANDISII as inputs for LANDIS-II modeling. Growing degree day calculation for all plants (baseline 5.56 degrees C) for modeled year zero and year 100 based on maximum and minimum temperatures for January and July (Botkin et al. 1972; Xu et al. 2009). Maximum growing degree days for *Pinus rigida* show an inability of this species to take advantage of higher temperatures in a changing climate for increasing biomass via photosynthesis as compared to maximum growing degree days for *Quercus alba* (GDDmax and GDDmin from Pastor and Post 1985).

GDD at Year		Species	GDDmin	GDDmax
0	100	*Pinus rigida*	1940	3100
3464.791	5342.297	*Quercus alba*	1721	5537

List of Figures:

Figure 1.

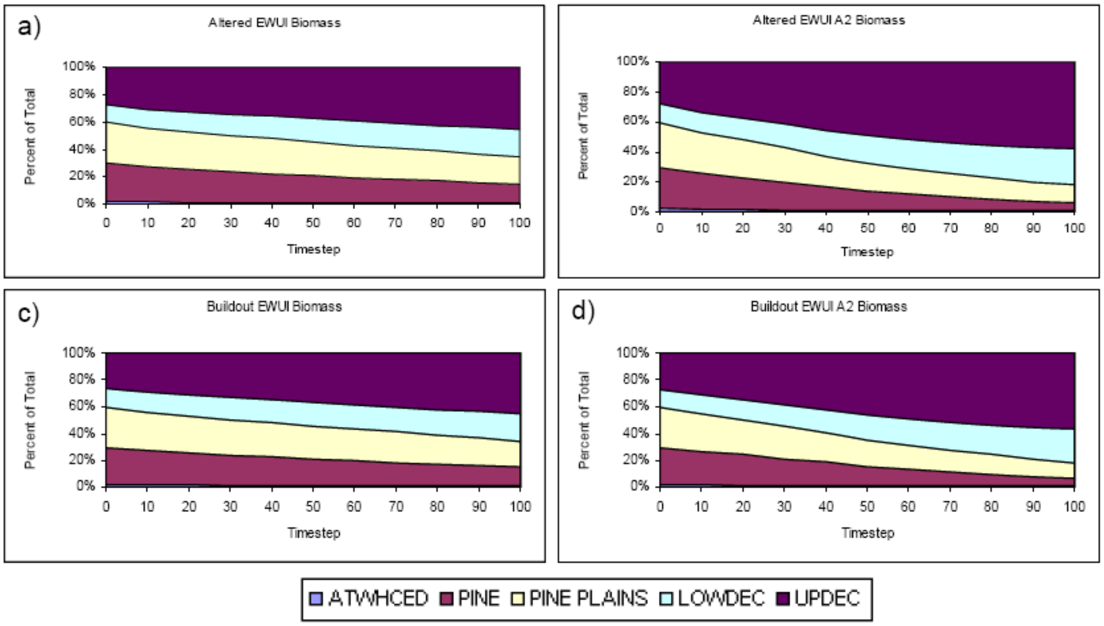

Figure 2.

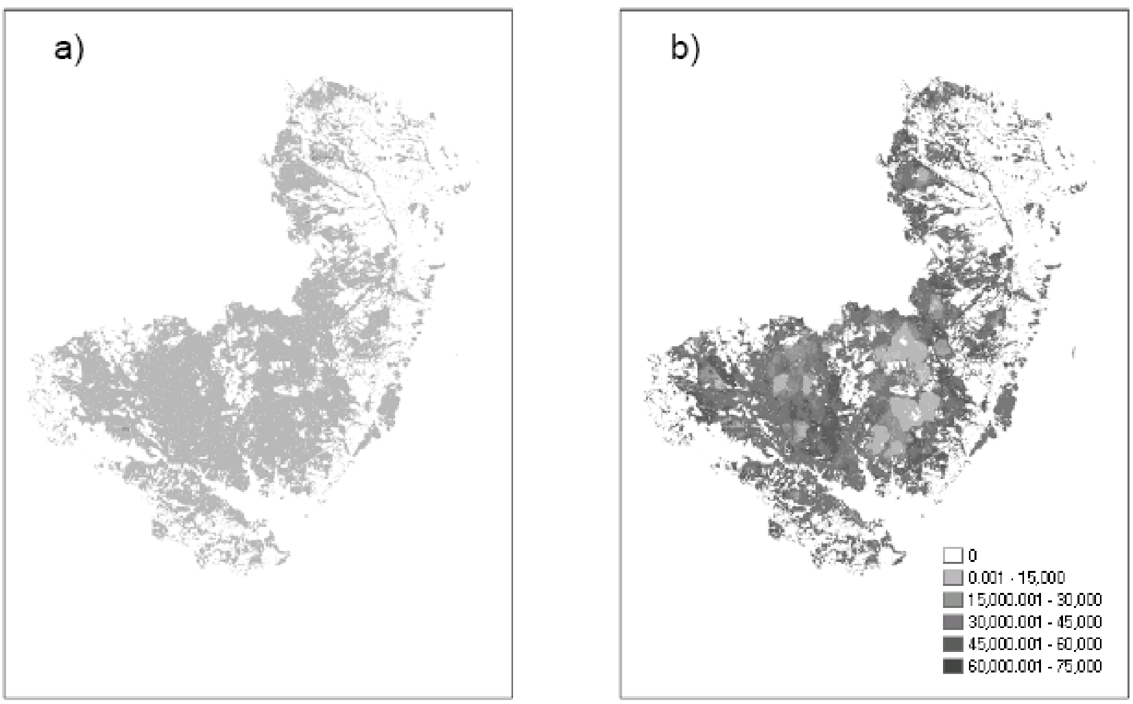

Figure 3.

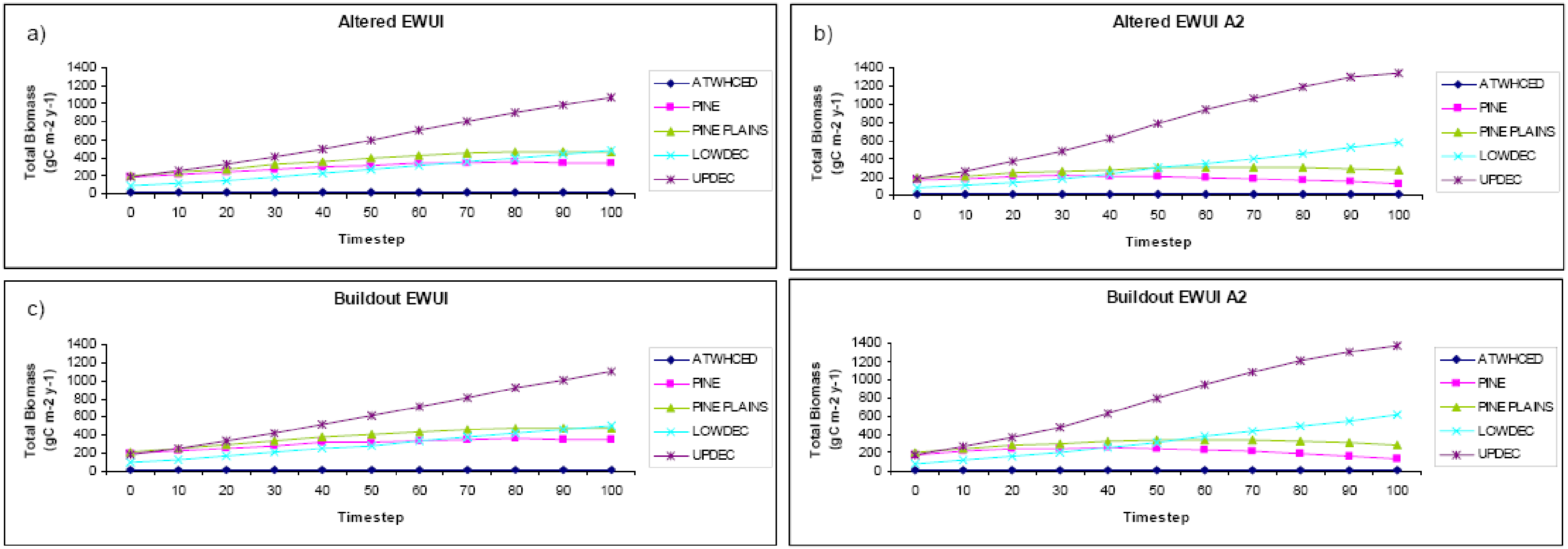
a)
Altered EWUI
b)
Altered EWUI A2
c)
Buildout EWUI
Buildout EWUI A2
Total Biomass (gC m-2 y-1)
Timestep
0
10
20
30
40
50
60
70
80
90
100
0
200
400
600
800
1000
1200
1400
ATWHCED
PINE
PINE PLAINS
LOWDEC
UPDEC

Figure 4.

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Concluding Remarks

My research has shown that the NJ Pinelands has been shaped by human activity during the last two centuries. In turn the Pinelands have shaped human activities both positively and negatively. This dissertation concentrated primarily on how humans have shaped the Pinelands ecosystem, but the Pinelands have also shaped human actions via abiotic and biotic influences. These influences have determined historic development trends such as cranberry and blueberry farming due to acidic and sandy soils; utilization of coveted resources, such as trees for charcoal and bog iron for tools; locations of military sites in ‘barren’ inhospitable lands, and the locations of beach developments with clean plentiful water supplied by the aquifers. My research attempts to assess more recent trends and forecast possible future changes on the human-environment interaction by providing insight into the effects of humans on the environment via investigating the effects of human alterations of the landscape on wildfire frequency and how these disturbances influence water quality and forest composition.

In creating an 80 year spatial fire history and using it to investigate and model the effects of changing fire regimes and altered land on forest composition in the Pinelands of New Jersey, I found numerous common threads. Overall, actual forest composition has changed from pine to oak cover over a twenty year time span in areas close to altered land that have experienced infrequent historical fire. Although the ecological wildland urban interface EWUI areas, or the areas with reduced fire frequency due to suppression near altered land resulting in forest composition change differ in scale and influence in the Barnegat and Mullica watershed, the direction of succession in the absence of fire is clear. The area of influence that humans have can be quite extensive (up to 480 m on average in Barnegat bay). This means that without intervention, this traditionally pine forest will eventually succeed into oak dominated forest. The question for stakeholders and managers is as complex as it is simple; what do we want, pine or oak? Do we want pre-colonial conditions that potentially had more oak cover (Kurczewski and Boyle 2000); and if so, how do we determine what that level of cover might be with little to no palynological data for our area (Watts 1979)? Do we take into account that almost all wildfires over the last century in the Pinelands have been human caused, which may distort our image of what the Pinelands should or should not be? Or do we want mostly pines, the modern version of forest cover for which this area was protected as a unique habitat? Which is better for ecosystem services such as fire safety, water quality, and carbon sequestration? Which is better for preserving endemic and endangered species habitat? Which is more viable in the face of climate change? What I can conclude from my research is detailed below via several management recommendations.

Management recommendation #1: Limit further development to minimize the expansion of the ecological wildland urban interface (EWUI).

The Pinelands commission and local municipalities are charged with enforcing zoning restrictions and updating the comprehensive management plan for the Pinelands National Reserve. It is clear from my research on defining the EWUI that new changes in human-altered land effectively reduce the area available for historic disturbance processes such as fire and cause succession from pine to oak cover. From the human perspective, this transition needs to be assessed for fire safety benefits or drawbacks, such as fuel moisture differences in broadleaf versus needleleaf leaves, typical ladder fuels in each cover type, and wildland firefighter access to pine versus oak dominated forest. The effects of gypsy moths on oak mortality may increase fire danger near life and property as well, although the southern pine beetle could create similar conditions in the future for pine dominated forest. From the ecosystem perspective, habitat change affects endemic pinelands species such as the pine snake and the numerous plants that rely on fire disturbance to persist. Additionally, the ability for the forest to sequester carbon (pine or oak) is reduced significantly with additional development based on loss of forest cover with development. Overall, I conclude that although there may be economic benefits to some by expanding altered land in the EWUI, these benefits are outweighed by reducing the capacity for the Pinelands to provide the essential ecosystem services to society as a whole.

Management recommendation #2: Determine more areas appropriate for prescribed fire and prepare for wildfire in areas not undergoing prescribed fire. If fire is a desired disturbance process in order to maintain the “unique ecological resources” and “the visual and ecological character” of pinelands vegetation, as specified in the Pinelands Comprehensive Management Plan (Pinelands Commission 1981, revised 2010), then fire must become more frequent via prescribed fire and/or ecologically based prescribed fire. Modeling of the modern fire regime from the last twenty years of the fire history showed that if the fire regime continues in this manner, wildfire will become smaller and less frequent due to altered land configurations (even with climate change) and will not sufficiently preserve pine cover except in very small swathes of its current range. It is possible that with focused low intensity prescribed fire and/or mechanical thinning either in the EWUI or adjacent to the EWUI, wildfire in interior natural areas can become more frequent and less intense, and the policy of complete suppression can be revised. My maps of fire frequency and the fire history database can be used to pinpoint areas in need of prescribed burning as well as areas where large fires have not occurred in some time and which may be prone to high intensity fire. Historically, prescribed burns were conducted for the creation of wildlife habitat and defensible fire lines. These areas were easier to burn due to the fire rotation interval maintained with institutional knowledge. Although the difficulty of conducting prescribed fires in new areas of the Pinelands is evident due to extensive fuel buildup and the expansion of altered land, and the efforts that the NJFFS must take to ensure public safety is significant, I feel that areas undergoing prescribed fire need to be expanded and locations for planned prescribed burns updated in order for this type of fire to be successful for ecological as well as safety purposes in the future. It may be necessary to reduce biomass via mechanical means in advance of prescribed fire to attempt to mimic wildfire with more severe growing season and heterogeneous fires, but severity recommendations for prescribed fire were outside the scope of this research.

Management recommendation #3: Incorporate climate change into management decisions.

The risks of ignoring climate change are evident in Chapter 4 forecasts of forest composition. Small decreases in the fire regime should be expected and the ability for oaks to outcompete pines in a CO_2 enriched, warmer environment will outweigh any fire effects. Pine cover will decrease to small islands of pine in a matrix of oak cover. Although CO_2 concentrations in the atmosphere and temperature increases can’t be controlled at the state level, limiting development of forests and managing for carbon sequestration begins at the local level. Policies developed in New Jersey can be models for other highly urbanized states containing significant forest resources. Increasing development densities using urban growth limits and investing in already altered land are some ways to address forest fragmentation (Radeloff et al. 2010).

Limitations

In conducting this research several interpretive limitations to the data and model became evident. In Chapter 1, limitations to fire history included a lack of reliable prescribed fire perimeters and information on severity for wildfires. In many cases, prescribed fire plans were clear, but areas that actually experienced burning were unclear in Division B records from the NJFFS. For Division C, no maps of prescribed fire were available at all. General statistics of areas burned per year was available. If prescribed fire is to be used in the future for ecological purposes, it will be critical to improve these records in order to understand long term forest processes in response to prescribed fire. Additionally, including severity in wild and prescribed fire history records will add significantly to the breadth of studies that could utilize such records.

In Chapter 2, water quality samples were inadequate to determine whether fire significantly affected pH, specific conductance, and turbidity or any nutrient of interest to ecosystem processes. The network of regular water samples should be expanded, intervals increased, and flow data added to all stations. Long term trends and flow relationships should be established for those areas without pulses of disturbance. Samples after fire should be increased over the short term and monitored for a return to pre-fire conditions over the long term. This monitoring should occur not only at the sampling station directly downstream of fire, but also at stations further along the stream network and in the bays and estuaries.

In Chapter 3, modeling the EWUI area as a step-change at 400 meters for Barnegat Bay or 200 meters for Mullica River rather a more desirable continuous gradient of change from the edge of altered land was a necessary drawback of modeling the area as a stand alone fire region. The difficulty of modeling intricate fire ecoregions in LANDISII prevented specifying more detailed differences in fire regimes at increasing distances from altered land. More extensive fire calibration and/or the improvement of fire modeling in LANDISII may help in this respect.

In Chapter 4, the limitations of LANDISII with respect to modeling climate extremes and epicormic shoot regrowth should be noted. Climate change studies have shown that climate change will not only come in the form of increased temperatures and CO_2, but also as increased precipitation or droughts and heat waves or cold spells (Gleason et al. 2008). New extensions and updates of LANDISII are addressing these problems and future research can incorporate these important ecosystem processes.

Final comments

The Pinelands of New Jersey represent a unique modern ecosystem and is home to numerous endemic fire dependent organisms. It seems likely that management of this area will be focused on preserving the wildfire and pine characteristics of this ecosystem in order to maintain this unique ecosystem for future generations. However, the results of landscape disturbance and succession modeling show that a serious effort in reducing altered land development and increasing fire frequency is needed to maintain the integrity of the Pinelands ecosystem. The possibility of maintaining current pine cover seems difficult although further modeling is needed to understand thresholds at which fire or other means of pinelands preservation are needed to maintain pine dominated forest. The current configuration of altered land may preclude the use of prescribed fire in maintaining pine cover and other methods may be more viable. Currently, the NJFFS is testing various fuel treatments to mimic fire in the wildland urban interface. Although not addressed in this dissertation due to my focus on the range of dominant forest types, mixed forest is also a major part of the pinelands and could be the predominant forest cover for vast areas of the landscape before either burning and reverting to pine or succeeding toward oak cover.

Overall, the results of this research need to be addressed in the context of the coupled human-environment system. Any course that we choose will have long term effects and may have unexpected consequences (e.g. a stable fire regime in a warming climate). Decisions made today in pinelands management will determine the resilience of the forest to future human-caused disturbances; therefore, mimicking known fire regimes and preserving current forest composition may be the best course of action.

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