# 10 years of change in the absence of fire: the Long Island Pine Barrens

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## Abstract:

Pine barrens are a globally rare ecosystem characterized by dry, sandy soils, and the presence of pitch pine (*Pinus rigida*), oak (*Quercus spp*.), grasses (*Carex spp*.), and heath species such as huckleberries (*Gaylussacia spp*.) and blueberries (*Vaccinium spp*.). Generally forming open, early successional habitats home to a variety of species, the pine barrens are a community historically driven by fire. But with widespread fire suppression policies put in place over the past 50 years, the pine barrens have begun to change. This study takes place at Brookhaven National Laboratory (BNL), located within the Long Island Central Pine Barrens, and aims to detect changes in pine barrens forest health and composition over the past decade (2005-2015). Analyzing data collected from 13 forest health plots in 2005 and 2015, I found that over the decade canopy cover has increased, tree stems have decreased in number, DBH of trees has increased, and species richness and abundance of tree seedlings and understory plants have increased. These changes have all been statistically significant, and suggest that our forests are going through ecological succession and perhaps the process of mesophication. This research is necessary for the future adaptation of forest management plans at BNL to maintain and conserve pine barrens forest, and for the prevention of forest conversion to a hardwood-dominated, shade-tolerant ecosystem.

## Introduction:

The Pine Barrens are a globally rare ecosystem characterized by dry, sandy soils, and the presence of pitch pine (*Pinus rigida*), oak (*Quercus spp*.), grasses (*Carex spp*.), and heath species such as huckleberries (*Gaylussacia spp*.) and blueberries (*Vaccinium spp*.) (Jordan *et. al*. 2003, Reiners 1967). Generally forming open, early successional habitats home to a variety of species, the pine barrens are a community historically driven by fire. Though often perceived to be a dangerous or high risk disturbance, fires have been burning for millions of years, before humans were even on the planet (Perry *et. al*. 2008, Pausas and Keeley 2009). Fires are great for renewing, maintaining, and diversifying ecosystems, and since they don't always burn everything evenly, they create a patchy mosaic of different aged vegetation that provides habitat for an abundance of species, both plant and animal (Perry *et. al*. 2008).

Up until the past fifty years, fires burned extensively in the Long Island pine barrens, as well as pine barrens throughout the country (Jordan *et.al.* 2003, Li and Waller 2015, Massada *et. al*. 2009). With increased human development, fragmentation, and fire suppression over the past several decades, many pine barrens communities are suffering degradation and shifting their composition from pine dominated communities to closed canopy, oak-dominated systems (Jordan et. al. 2003, Kretchun *et. a*l. 2014, La Puma et. al. 2013, Bried *et. al.* 2014, Kurczewski and Boyle 2000). Fire suppression is deemed necessary by many forest managers to lower the risk of fire and its effects to humans and structures at the wildland-urban interface (WUI) (Massada *et. al*. 2009). However, suppression creates large buildups of volatile fuels that can become worrisome when not managed properly (Bried *et. al.* 2014). Presence of roads throughout these ecosystems can provide a buffer that reduces the continuity of fuels (Scheller *et. al*. 2008), but as development and fragmentation increase, using prescribed fire as a management tool becomes extremely limited. Lack of forest management through fire and increased edge effects due to human development can accelerate succession to form a more oak-dominated system, especially on forest edges and within the WUI. Disturbances, such as fire, weather events, and human development are events that alter the biological imprints on a site, shifting the ecosystem to earlier stages of succession. Succession is a progressive and gradual change in the composition of a community following a disturbance (Perry *et.al.* 2008). The advantage to this type of succession close to development is that although ignitions and fires are more common within the WUI, oak species create more shaded and moist microenvironments that decrease fire severity and size (La Puma *et. al*. 2013, Massada *et. al.* 2009). However, this shift not only threatens to eliminate many of the rare plants and animals dependent on the pine barrens ecosystem, but to prevent the pine barrens ecosystem from regenerating at all (Howard *et. al*. 2011, Howard 2015, Jordan *et. al*. 2003).

In the 19th century, the pine barrens in Suffolk County, Long Island, NY were the second largest pine barrens system in the northeastern US (Kurczewski and Boyle 2000). But with fire suppression and human development in the 20th century, about half of this system has been lost, and what remains has been converted to oak-hardwood, pine-oak, and oak-pine forests (Jordan *et. al*. 2003, Kurczewski and Boyle 2000). This study takes place at Brookhaven National Laboratory (BNL), which is located within the Long Island Central Pine Barrens. It aims to detect changes in pine barrens forest health and composition over the past decade (2005-2015) by looking at significant changes in tree abundance and size, canopy cover, abundance and diversity of seedlings, and understory diversity and abundance.

## Methods

### Study Site

BNL is a Department of Energy research facility that sits on 2,130 hectares of land in Upton, NY in Suffolk County, Long Island. In 1917, the land was cleared for the creation of Camp Upton, a United States Army base that served to house troops during both World Wars (Brookhaven National Lab). In the time between the World War I and II it was replanted as Upton National Forest by the Civilian Conservation Corps. In 1947, Camp Upton became Brookhaven National Laboratory, dedicated to research on atomic energy (Brookhaven National Laboratory n.d.).

Currently, BNL is part of the 41,480 hectares of protected pine barrens forest on Long Island, one of only a few pine barrens ecosystems in the country. Its forests are characterized by pitch pine, oak species, and an understory consisting of scrub oak (*Quercus ilicifolia*) and various heath species such as blueberries and huckleberries (Brookhaven National Laboratory 2011). Plots within this study were located within four different forest types: coastal oak forest, pitch pine forest, pitch pine-oak forest, and oak-pine forest. Coastal oak forests can have varying communities, from tree oaks mixed with hickory and a diverse shrub layer, to an oak-dominated canopy with a continuous shrub layer of huckleberry and blueberry. These forests have less than 10% pitch pine cover. Pitch pine forests have greater than 90% pitch pine cover, with a continuous shrub layer of huckleberry, blueberry, and scattered scrub oak. Pitch pine-oak forests have a canopy comprised of both pitch pines and oaks, with pitch pines accounting for 51-90% of the cover. This forest type also has a nearly continuous shrub layer of huckleberry and blueberry with scattered scrub oak. Finally, oak-pine forests also share a canopy of pitch pine and oak, with tree oaks accounting for 51-90% of the canopy cover. This forest type also has a shrub layer with huckleberry, blueberry, and scattered scrub oak (Foundation for Ecological Research in the Northeast 2007).

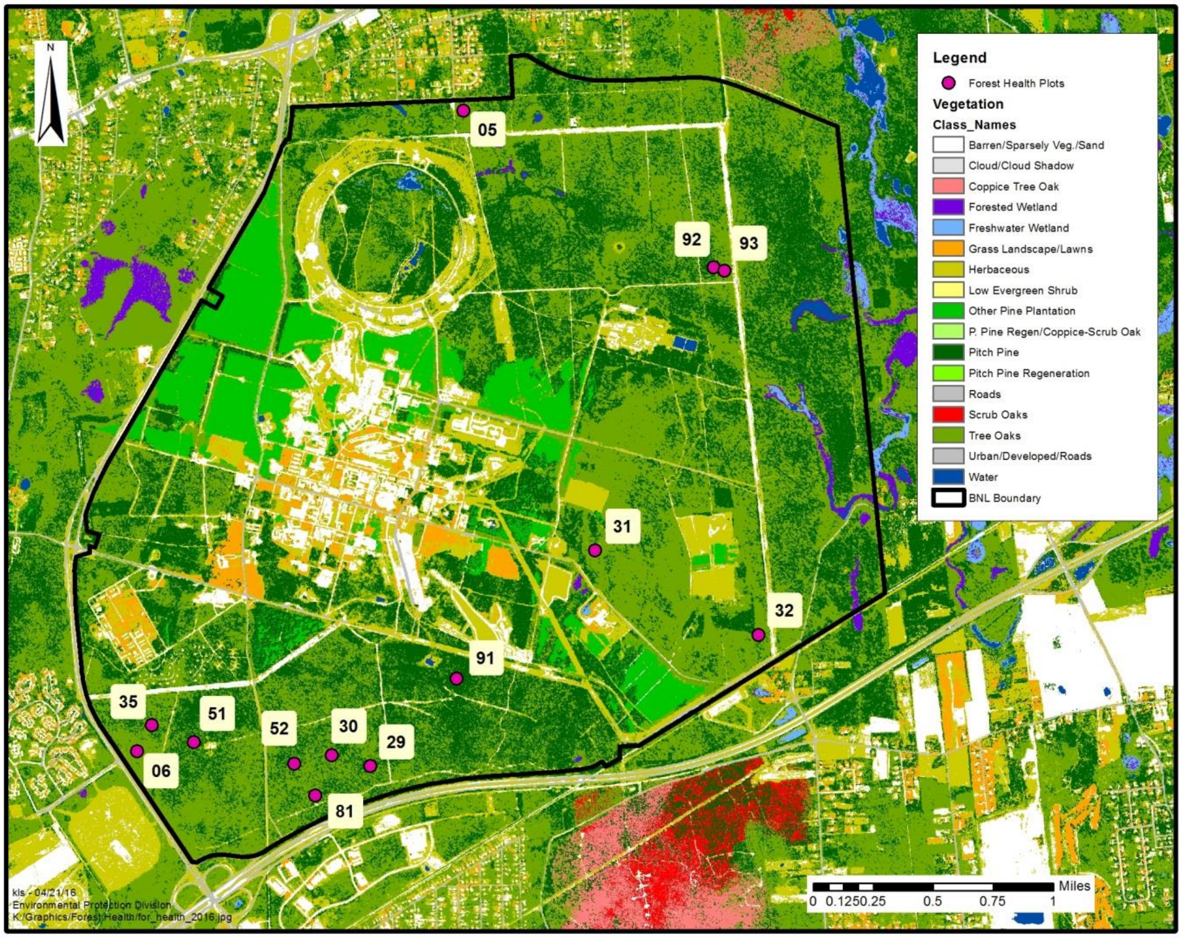


**Figure 1: Thirteen forest health plots located in pitch pine, oak-pine, pine-oak, and coastal oak forest types. Surveyed in both 2005 and 2015 at Brookhaven National Laboratory, Long Island, NY**

## Forest Health Monitoring Protocols

In both 2005 and 2015, thirteen forest health plots within BNL were surveyed as a part of a larger Central Pine Barrens forest health monitoring project that was created with the intention of assessing and calculating forest growth and change over a ten year period. Plots surveyed were located in oak-pine, pine-oak, pitch pine, and coastal oak forest types. Dominant plants found throughout the plots included pitch pine, white oak (*Quercus alba*), scarlet oak (*Quercus* *coccinea*), scrub oak, black huckleberry (*Gaylussacia baccata*), late lowbush blueberry (*Vaccinium pallidum*), and early lowbush blueberry (*Vaccinium angustifolium*).

To establish plots within BNL, coordinates were chosen randomly using ArcGIS™. Within each 16 x 25 meter plot, all flora and fauna species were recorded, photographs were taken, strata percent cover and height were recorded, and ten transects were run along the 16 meter edge (Foundation for Ecological Research in the Northeast 2007). Starting points on each transect were randomly chosen, and a 2 meter tent pole was placed and used to record all adjacent plant species at the starting point, and at each additional meter along transects. Litter and duff samples were taken at four locations along transects, and canopy cover type (pitch pine and/or hardwood) was also noted. Four belt transects were constructed within each plot to collect data on seedling and sapling abundance. Once 10 seedlings of a given species were located within a belt transect, we stopped recording for that species. Live trees, dead trees, and downed logs greater than 10-centimeters in diameter at breast height (DBH) were also identified and measured for each plot.

## Statistical Analyses

As the data collected for this study proved to be non-parametric, Wilcoxon signed rank tests (Social Science Statistics 2016) were run on all aspects of the data to determine significant changes in forest composition over the past decade. The data analyzed included canopy cover, mean DBH of trees greater than 10cm, total number of tree stems greater than10cm DBH, tree seedling abundance and richness, and understory species richness and abundance. Percent change in these key ecological attributes of the forest over a 10 year period of time was also calculated as an additional determination of how much the forest has changed over this period of time.

## Results:

### Canopy Cover:

In 2005, hardwoods were the predominant canopy cover with 61% of data points listed as having hardwoods above them. The amount of hardwood cover in the canopy increased significantly (p-value = 0.01732) from 2005 to 2015, about 25% overall, and remained the dominant canopy cover in 2015. Pitch pine was only noted in 23% of data points in 2005, and also increased significantly (p-value = 0.00338) over the decade with a 62% increase in cover overall. Although the make-up of the canopy changed from 2005-2015, with hardwood and pitch pine cover increased in 2015, the overall percent canopy cover averaged across plots did not change significantly over time (p-value = 0.89656). However, the percent sub-canopy cover increased significantly (p-value = 0.00188), across plots averaging 1.23% in 2005, and 30.15% in 2015.

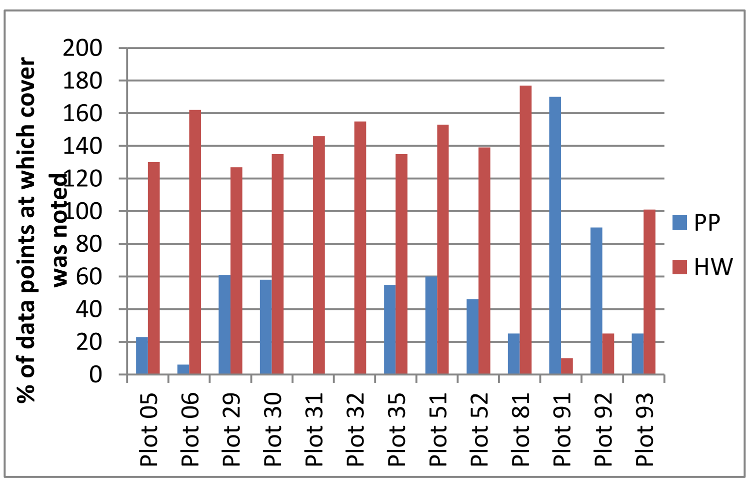


**Figure 2: Pitch pine cover abundance versus hardwood cover abundance across forest health plots at Brookhaven National Laboratory, surveyed in 2005**

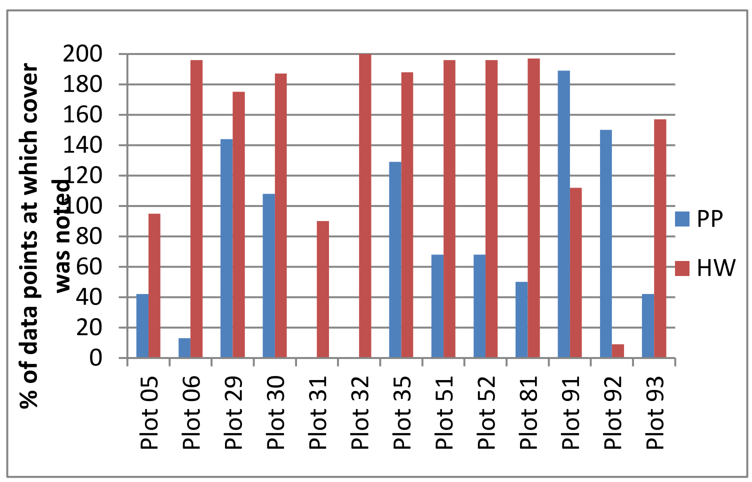


**Figure 3: Pitch pine cover abundance versus hardwood cover abundance across forest health plots at Brookhaven national laboratory, surveyed in 2015:**

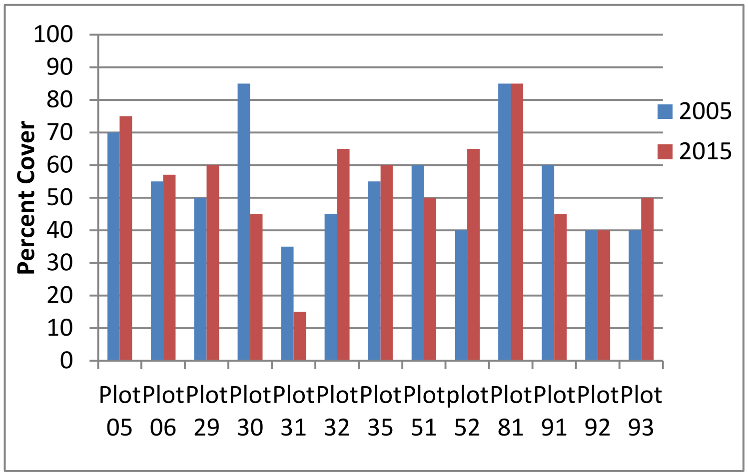


**Figure 4: Percent canopy cover across forest health plots in 2005 versus 2015 at Brookhaven national Laboratory**

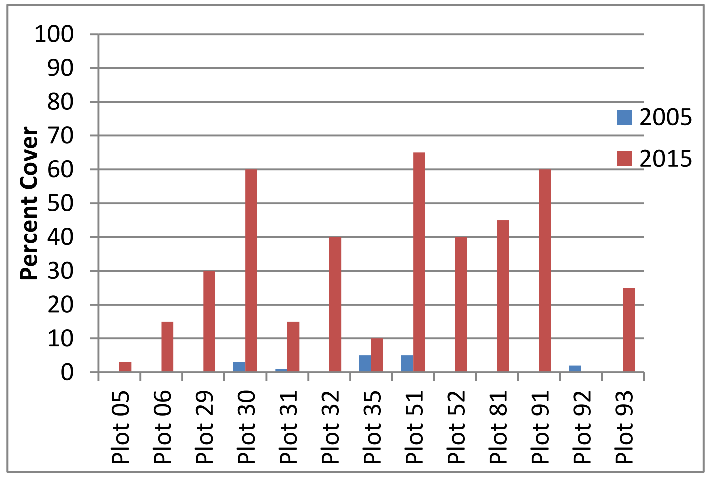


**Figure 5: Percent sub-canopy cover across forest health plots in 2005 versus 2015 at Brookhaven National Laboratory**

## Trees:

For ease of analysis, when looking at mean DBH and total number of stems for trees greater than 10cm DBH, only the two most predominant categories of trees – pitch pines and oak species – were analyzed. The DBH for each category was averaged for each plot so that plots could be more easily compared to each other from year to year. For both categories of trees, pitch pines and oaks, the total number of stems and the mean DBH changed significantly from 2005 to 2015. For each category, the total number of stems decreased over the past decade, while the mean DBH increased. These changes were all calculated to be significant (p-value = 0.020879 for total oak stems; 0.0394 for oak mean DBH; 0.029974 for pitch pine stems; and 0.00338 for pitch pine mean DBH), with oak stems decreasing in number more than pitch pine stems (21.28% decrease versus 15.58% decrease), and pitch pines increasing their mean DBH more than oaks (16.7% increase versus 6.9% increase).

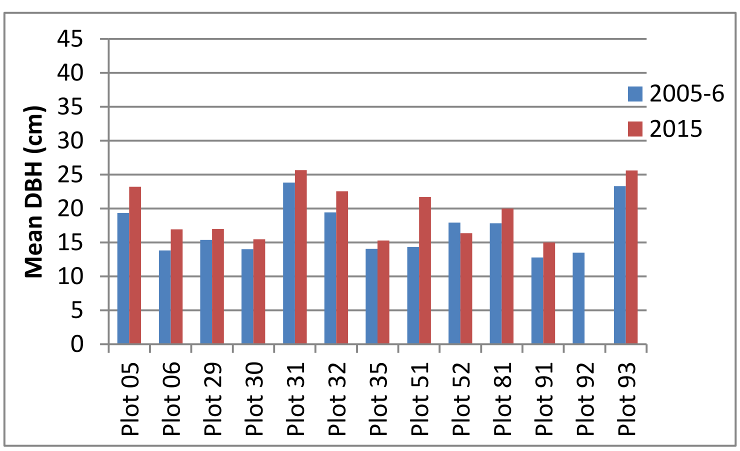


**Figure 6: Mean DBH averaged within plots for all oak species in 2005 versus 2015 at Brookhaven National Laboratory**

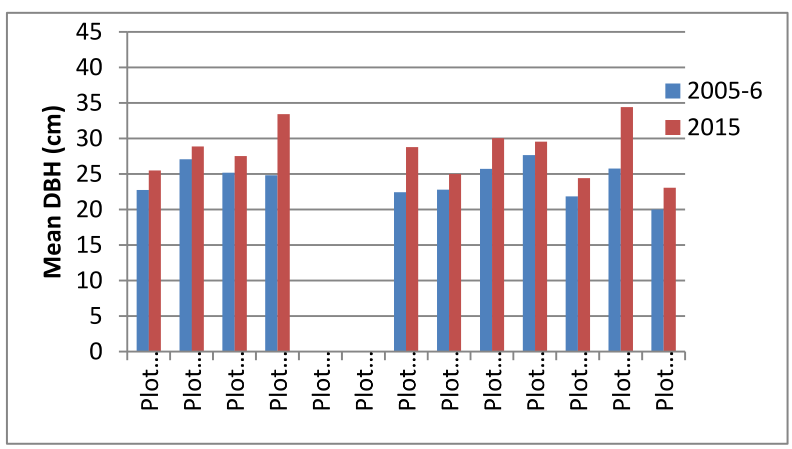


**Figure 7: Mean DBH averaged within plots for all pitch pines in 2005 and 2015 at Brookhaven National Laboratory**

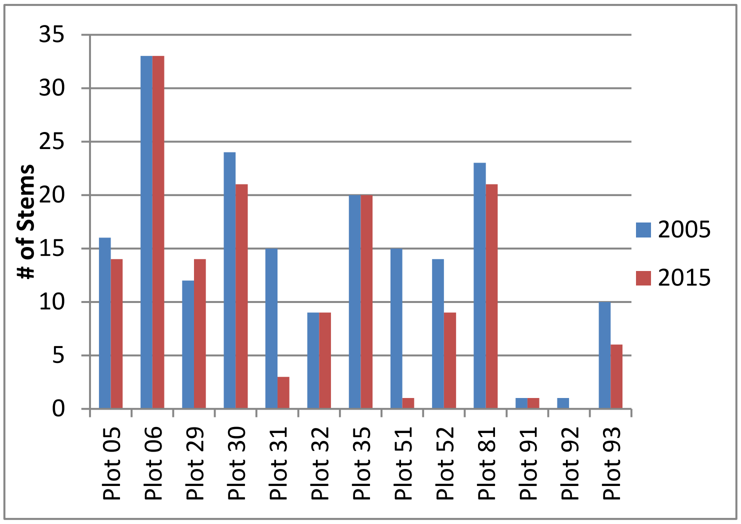


**Figure 8: Total number of stems >10cm DBH for all oak species in 2005 versus 2015, at Brookhaven National Laboratory**

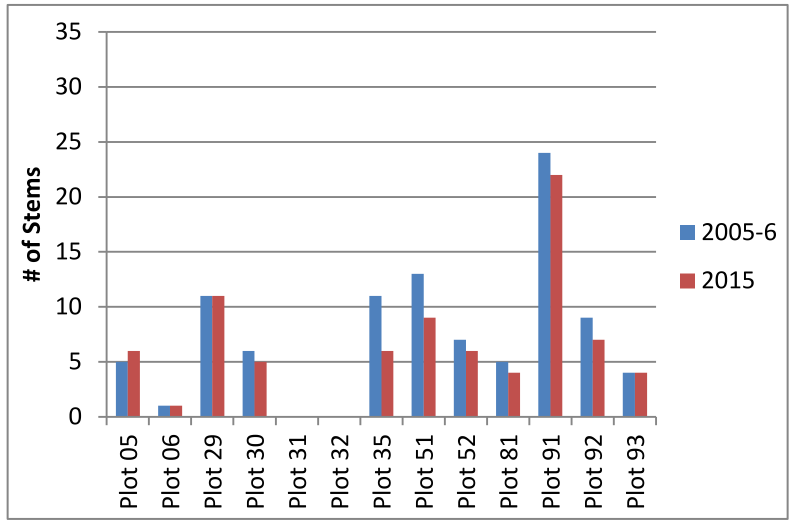


**Figure 9: Total number of stems >10cm DBH for all pitch pines in 2005 and 2015, at Brookhaven National Laboratory**

## Tree Seedlings:

Tree seedlings also changed significantly in abundance (p-value = 0.00148) and species richness (p-value = 0.00338) from 2005-2015. Number of seedlings across the 13 forest health plots increased from 301+ seedlings recorded in 2005 to 1,039+ seedlings recorded in 2015. Oak and pitch pine seedlings alone also increased significantly over the decade (p-value = 0.00222 and 0.020863, respectively), with about 2.5 times more oak seedlings, and about 6 times more pitch pine seedlings recorded in 2015. The number of species present in each plot was averaged across all plots for both 2005 and 2015, and showed that seedling species richness across the study site has more than doubled over time. In 2005, there was on average 2.15 species of tree seedlings found in each plot, and in 2015, the average number of species of tree seedlings increased to 4.54. The individual plots with the highest seedling richness in both years was plot 32 in 2005 with 4 species of tree seedlings present, and plots 31 and 32 in 2015 with 8 species of tree seedlings present. In 2005, only 5 species of tree seedlings were present across all 13 plots, and plot 32, a coastal oak forest type, was the only plot that had tree seedlings other than pitch pine or oak recorded. In contrast, 8 species of tree seedlings were present across the plots in

2015, and non-oak hardwood tree seedlings were found in every plot except for plot 93, which is a pitch pine forest type.

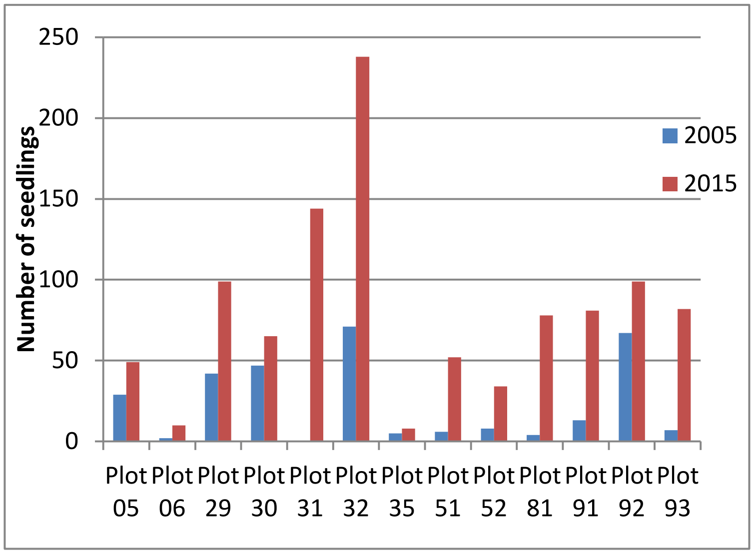


**Figure 10: Number of tree seedlings recorded in 2005 versus 2015 at Brookhaven National Laboratory**

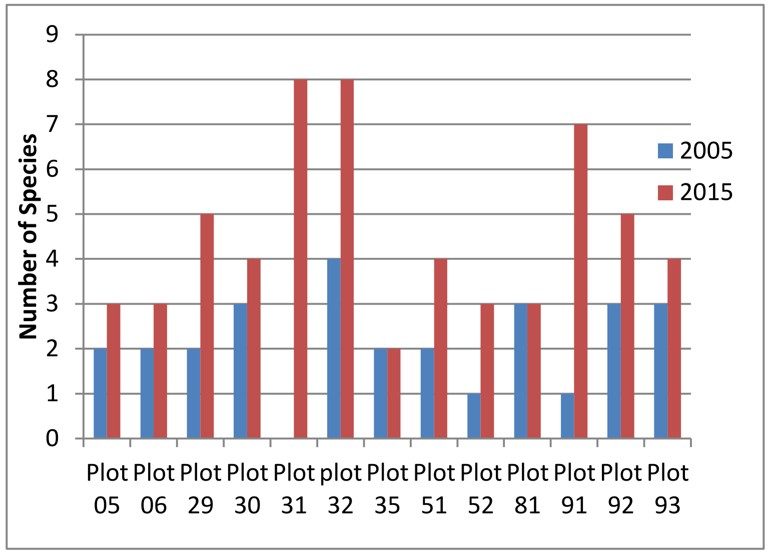


**Figure 11: Tree seedlings species richness in 2005 versus 2015 at Brookhaven Nationals Laboratory**

## Understory:

Due to the large amount of species located within the understories of the forest health plots surveyed, only four of the most predominant species were analyzed to look at differences in abundance of species over the decade. These species included black huckleberry, early lowbush blueberry, late lowbush blueberry, and Pennsylvania sedge (*Carex pensylvanica*), and were found to have not changed significantly from 2005 to 2015 (p-value = 0.87288 for black huckleberry, 0.53526 for early lowbush blueberry, 0.75656 for late lowbush blueberry, and 0.65994 for Pennsylvania sedge). However, species richness in the understory doubled from 2005 to 2015 (p-value = 0.00714) with only 13 species present across the study site in 2005, and 26 species present in 2015. If we take out the species most common to the pine barrens understory (i.e. huckleberries, blueberries, oaks, pitch pine, Pennsylvania sedge, and bracken fern (*Pteridium aquilinum*)), we are left with 15 additional species present in the 2015 understory.

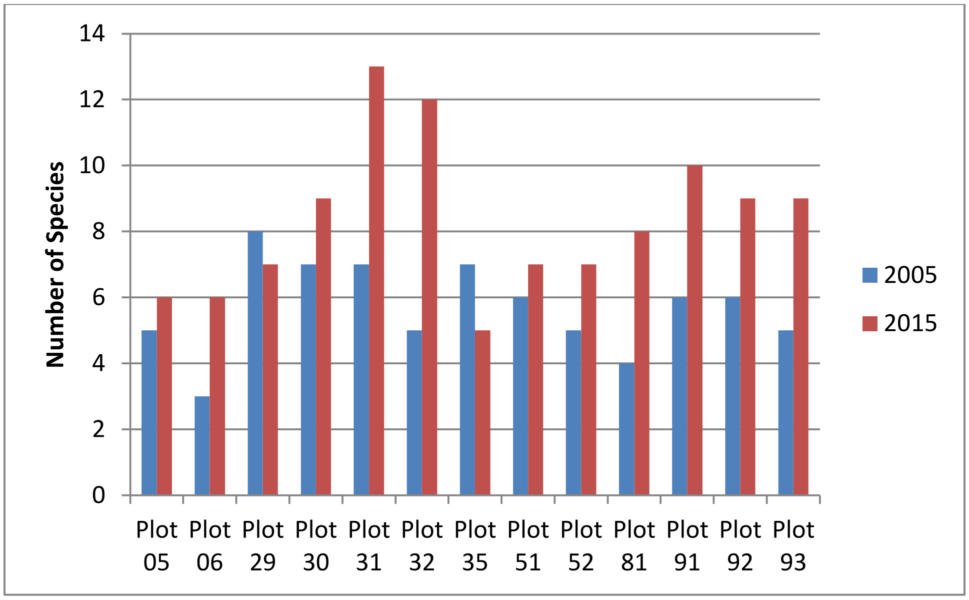


**Figure 12: Understory species richness in 2005 versus 2015 at Brookhaven National Laboratory**

## Discussion:

With widespread policies of fire suppression in the United States over the past several decades, we have been able to get an idea of how pine barrens ecosystems grow and change in the absence of fire (Jordan *et. al*. 2003, Kurczewski and Boyle 2000, Howard 2015, Olson 2011). It has been generally noted that as fire becomes less common within this system, preventing it from properly regenerating, hardwood trees begin to invade, reducing the openness of the system by forming closed canopies that block out light (Bried *et. al*. 2014, Jordan et. al. 2003, Kurczewski and Boyle 2000, Howard 2015, Howard *et. al.* 2011, Kretchun *et. al.* 2014). This development of closed canopy forest affects light levels within the forest, composition and depth of leaf litter, coarse woody debris, moisture levels, and vegetation composition, eventually leading to replacement of all of the factors that make the pine barrens unique.

Jordan *et. al.* (2003) looked at ecological models for the Long Island pine barrens, and the management implications that came from them. Looking at the effects and results of fire at increasing intervals of longevity, they noted that after 10-20 years of fire suppression, oak canopies begin to close in, reducing the amount of sunlight in the understory. Within BNL, we found that hardwood canopy cover was dominant over pitch pine cover in both 2005 and 2015, and that although pitch pine cover had a much greater increase over time than hardwood cover, hardwood cover still increased by about 25%. However, the fact that hardwood cover has been significantly increasing over the past decade, and that our trees have been significantly decreasing in number of total stems and increasing significantly in DBH leads us to believe that our forests are indeed going through succession, and possibly through mesophication (Nowacki and Abrams 2008).

According to Nowacki and Abrams (2008), mesophication is “the escalation of mesic micro-environmental conditions, accompanied by ever diminishing prospects for fire-dependent species.” Essentially, in the absence of fire, canopies will close causing fire-adapted species to perform poorly in the lower light conditions, and eventually give way to shade-tolerant, less fire-tolerant species that promote moist, cool, microenvironments (Nowacki and Abrams 2008, Howard 2015). In a study in the New Jersey pine barrens, Olson (2011) found that when comparing burned and unburned forest stands, there was higher non-oak hardwood diversity and higher mean total seedling abundance in unburned stands. In our forest plots, which are all considered unburned, we noticed that tree seedling abundance increased significantly, by about 245%, and that tree seedling species richness also increased significantly, by about 111%. In 2005, five species of tree seedlings were found: black oak species, white oak, pitch pine, red maple (*Acer rubrum*), and sassafras (*Sassafras albidum*). Out of all 13 plots surveyed, plot 32 was the only one with non-oak hardwood species present. In 2015, three additional non-oak hardwood species were noted as seedlings: black cherry (*Prunus serotina*), serviceberry

(*Amelanchier canadensis*), and blackgum (*Nyssa sylvatica*). In this year, all plots but plot 93 (pitch pine forest type) had these non-oak hardwood species present. All five of the non-oak hardwood species found throughout our plots can be considered dry mesic to mesic species (USDA Forest Service 2016, USDA Natural Resources Conservation Service 2016), and many of them are shade tolerant, meaning that as they age and grow, they will promote a cooler, moist understory, and change the composition of the litter layer below – altering flammability of the litter and creating a seedbed more suitable to shade-tolerant species (Nowacki and Abrams 2008).

Factors that affect the dispersal and establishment of seeds also affect the competitive status of that plant species within its community (Denslow 1980). The factors that result from oaks and other hardwoods forming a closed canopy, such as decreased sunlight and increased duff and litter levels, negatively influence the establishment and growth of pitch pine seedlings, which are unable to penetrate thick leaf litter and need high levels of sunlight to be successful (Howard 2015), and favor other more shade tolerant species. Pitch pines are a dominant species within the pine barrens ecosystem, allowing light to filter through to the forest floor, and remaining a strong component of the forest after major fires. They have several adaptations and positive responses to fire, including cone serotiny, bole and crown sprouting, thick, heat resistant bark, dead branch retention, and germination of seeds on exposed mineral soil (Bond and Keeley 2005, Howard 2015, FEIS). These pitch pines can remain standing for 200 years or more, and remain a part of the pine barrens ecosystem, but over time if seedlings are unable to germinate, re-sprouting ability may decrease in aging individuals, and the seedbed could lose its viability (Jordan et. al. 2003, Howard 2015, Nowacki and Abrams 2008). However, our study found that within our plots, pitch pine seedlings increased significantly from 2005 to 2015 with an average of one seedling per plot in 2005, and eight seedlings per plot in 2015. This increase goes against what was expected with the succession of the pine barrens towards a more hardwood dominated forest, but just because seedlings were noted does not mean that they will grow to reach the sapling stage before getting shaded out by other plants.

The issue with mesophication of our pine barrens forest lies not only in the loss of a rare ecosystem and the unique wildlife that come with it, but in the idea that as the forest succeeds towards a more shade-tolerant, mesic condition, we actually lose the ability to reset the ecosystem with fire (Nowacki and Abrams 2008). After oaks come in and dominate the forest, the fire-resistant hickory (*Carya spp*.) and American beech (*Fagus grandifolia*) begin to invade. Downed woody debris and leaf litter become more compact, making it more difficult for fire to move through, and volatile species such as the blueberry and huckleberry found in our present-day understory may not be present in the future due to lack of sunlight in the understory.

The pine barrens forest throughout Long Island is unhealthy for a number of reasons. In addition to development, fragmentation, and fire suppression, our forests are negatively impacted by an overabundant white-tailed deer (*Odocoileus virginianus*) population that impedes the regeneration of both woody and herbaceous species, thereby reducing species diversity in the understory. Lack of active forest management has also resulted in forests that are overstocked, which has made trees more susceptible to insect damage from the southern pine beetle (*Dendroctonus frontalis*), orange striped oakworm (*Anisota senatoria*), and gypsy moth (*Lymantria dispar dispar*) (Nowak *et. al*. 2015, Showalter and Turchin 1993). Future ecological management for the pine barrens should focus on preserving and maintaining a patchy mosaic of forest types to maximize habitat for a large variety of plants and wildlife. This can be managed with mechanical treatments and prescribed fire. The use of mechanical treatments (using heavy equipment to thin the understory), and selective cutting of trees to reduce basal area in forest stands would be beneficial to the forest by alleviating stress caused by high competition, and by reducing the fuel load to help prevent prescribed burns and wildfires from moving into the canopy. Regardless, it is important to continue monitoring the forests, especially in conjunction with management, to continually assess changes in forest health and composition to ensure that management objectives are being met.

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