Paper Title:	*Understory Biodiversity Proportion Ground Cover and Available Soil Nutrients*
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Collection:	Biological Station, University of Michigan (UMBS)
Course:	EEB 381: General Ecology
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Date:	June 15, 2017
Abstract:	
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Signed,

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Understory Biodiversity, Proportion Ground Cover, and Available Soil Nutrients as

Functions of Time Since Major Burn in a Temperate Hardwood Forest

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Ecology 381

June 14, 2018

Abstract

Forest fires are major ecological disturbances that can drastically alter an ecosystem for many years (Gill et al. 1994). Through the process of secondary succession, a disturbed ecosystem is recolonized by species and can eventually reach its pre-burn levels of biodiversity (Moorhead 1996). While the return of biodiversity that accompanies secondary succession is well-documented (Moretti et al. 2006, Driscoll 2010, Granström 2001), the process by which understory species diversity and ground cover are restored over time, particularly in temperate hardwood forests, is less well-documented than in areas currently more prone to fires (Gill et al. 1994, Hanes 1971, Keeley 1981, Adler et al.1998). With ongoing climate change making naturally-occurring forest fires a more realized possibility in more northward regions (Handler et al. 2014), a better understanding of secondary succession in temperate hardwood forests may allow for more effective future ecological management of burned areas. This text utilizes data from controlled burn plots burned in 1911, 1936, 1980, and 2017 to provide a better understanding of this process; additional factors evaluated included impacts of light availability on understory ground cover and levels of soil nutrients based on time post-burn. No data showed any significant correlations between any of these parameters, except that the 2017 burn plot was found to vary significantly from the other burn plots in biodiversity, in terms of both species richness and evenness. This may be because the process of secondary succession tends to be complete by forty years after a burn (Moorhead 1996) and the three oldest burn plots sampled were close to or older than forty years post-burn. This timeline of succession, along with likely uneven burning of the plots, may help explain our lack of significant data. This reinforces the idea that the first forty years after a burn are the most important in terms of ecological management and restoration of understory biodiversity in burned areas. Furthermore, the lack of correlation between light availability and understory ground cover suggests that the restoration of canopy complexity is not necessarily an impediment to the recolonization of understory species, and that the two can coexist well in recovering temperate ecosystems.

Introduction

Naturally occurring forest fires are significant ecological disturbances that can drastically alter an ecosystem (Gill et al. 1994). Although a forest fire can leave a plot of land initially devoid of life and organic matter, a disturbed ecosystem can be recolonized and eventually return to its former level of biodiversity in terms of both species richness and evenness (Moorhead 1996). This occurs by a process known as secondary succession, and it can stretch for decades or even centuries before reaching its climax, at which there are stable levels of biodiversity within the community (Sahney 2008). Primary succession is the process by which a forest is initially colonized by species; secondary succession, however, describes the recolonization of a forest following a natural or human disturbance, such as a forest fire (Moorhead 1996).

While the return of biodiversity that accompanies secondary succession is well-documented (Moretti et al. 2006, Driscoll 2010, Granström 2001), the process by which diversity is restored specifically within the understory is less well-documented (Gill et al.1994). Furthermore, secondary succession in temperate hardwood forests is less well-documented than in forests in which naturally-occurring forest fires are more common, such as those in warmer, drier climates (Hanes 1971, Keeley 1981) or surrounding active volcanoes (Adler et al.1998). Additionally, increasing temperatures and more erratic precipitation patterns will likely occur in the Great Lakes region as a result of climate change (Handler et al. 2014). Accordingly, the species composition of the temperate hardwood forests of northern Michigan may be altered, but there will likely remain hardwood forests, cumulatively making the region more vulnerable to forest fires (Handler et al. 2014). Therefore, knowledge of secondary succession in temperate forests will become more relevant in many fields of ecological management.

Generally, the process of succession can be observed through changes in species composition, light, and soil nutrient availability in a habitat over time (Guariguata 2001). Thus, the observance of understory abundance and composition in burn plots of various ages may allow for the delineation of the rate at which understory cover and biodiversity is restored in a temperate hardwood forest following a major burn. Furthermore, evaluation of two major abiotic factors which may vary over the course of succession may be similarly valuable: light availability and the levels of various soil nutrients. Light availability in the understory generally decreases through the process of secondary succession with the increased canopy cover that comes with the recolonization of tree species (Guarigata 2001). Since this level of canopy cover is the largest determining factor in the amount of light availability in the understory, in this text the mean adult tree abundance and DBH (Diameter at Breast Height) are used as inverse proxies

for light availability to evaluate the impacts of light availability on understory ground cover. Additionally, burning a forest releases nutrients like accessible nitrogen and organic carbon, which is stored in organic material prior to decomposition or burning, and thus it is expected that more recently-burned plots will contain soils with higher levels of nutrients like available nitrogen and organic carbon (Nave et al. 2011)**.** As a result of these processes, it is expected that a steady increase in species biodiversity, in terms of both species evenness and richness, will occur immediately post-burn, in concert with the initial recolonization of species, before these values eventually become relatively constant (Moorhead 1996).

The burn plots used in this study are located in Pellston, MI on the the University of Michigan Biological Stations (UMBS) property. They were burned in 1936, 1980, and 2017. For use of the control plot, a geographically similar plot directly south of a 1998 burn plot was used; this area was not recently burned, but may have been burned along with the entire surrounding forest in 1911 (Nave et al. 2017). It is anticipated that biodiversity steadily increases with time after a major burn before reaching a relatively constant level (Moorhead 1996). In terms of ground cover in relation to abundance of adult trees (as an inverse proxy for light cover), it is anticipated that tree abundance would be inversely correlated with ground cover, as a greater number of adult trees will increase canopy density and allow less light to reach the understory, ultimately reducing understory ground cover (Thomas et al. 1999). Ground cover is defined in this paper as the cumulative length of all understory individuals lying along a given length of transect, including overlapping lengths of two or more different species. It is also anticipated that the mean DBH of trees and understory cover will be similarly inversely correlated, as mean DBH will be directly correlated with canopy cover, again reducing light availability and understory ground cover (Canham et al. 1994). It is anticipated that the levels of available nitrogen, organic

carbon, and phosphorous will be inversely correlated with length of time since burn, as burning

releases various nutrients from organic matter (Nave et al. 2011).

Materials and Methods

Materials

Transect tape, flag markers, DBH tape, meter stick, pen, data sheet, 10cm x 10cm wood square, knife, ball mill, 50 ml conical tube, 1 mm sieve, dryer (60°C oven), petri dishes, assimilation vials

Methods: Laying Transects

This project focused on the effects of the passage of time after a major burn on understory biodiversity and soil composition, as well as the impacts of light availability on amount of understory ground cover. The observed plots were burned in 1936, 1980, and 2017, and the control area was most likely burned in 1911 along with the rest of the surrounding forest (Nave et al. 2017). These areas represent a clear and tangible timeline of forest succession stretching back many decades, making them an ideal location for this topic of research.

In order to measure plant biodiversity of the burn plots, two 30-meter transects were laid in each plot. Flags were placed to mark the beginning and end of each transect and record its position for follow-up visits to the field. All of these transects began and ended at least 30 meters away from surrounding roads and other burn plots in order to minimize impacts of the edge effect. The transects were separated from one another by at least 20 meters and laid parallel to one another for consistency.

Methods: Biodiversity

The starting and stopping points of all understory plant species lying directly along the transect were measured and recorded. When different species overlapped one another along the transect, the starting and stopping points of each species were recorded in order to report the

ground cover and biodiversity of the understory as completely as possible. In order to compare species richness and evenness, each transect was split into two 15 meter samples, and the Shannon-Wiener and Simpson's Index were calculated for each sample. P_i was used to represent the total ground cover a species occupied along a given length of the transect, divided by the total ground cover for the same length of transect. The data for all of the samples were then grouped by burn plot, and normality testing was conducted in SPSS in order to ensure that the population of indices for each burn plot followed a normal distribution. If the data followed a normal distribution, ANOVA testing was conducted to compare the mean indices across the burn plots. Nonparametric Kruskal Wallis tests would have been used in order to compare means if the data was not normally distributed. Post-hoc Tukey tests were conducted following significant ANOVA results in order to pinpoint specific differences in the biodiversity indices between the separate plots.

Methods: Soils

In addition to studying changes in understory biodiversity between the plots, the levels of various soil nutrients for each of the burn plots were compared by taking soil samples along the transects. Soil samples were collected in the form of "brownies," which involved cutting a 10 cm by 10cm square from the A-horizon layer of the soil, and storing the samples in labeled 50 ml conical tubes. Three samples were taken from each transect at the 5 m, 15 m, and 25 m marks. After gathering the soil samples in the field, they were placed on petri dishes in a 60°C dryer for two days in order to dehydrate the samples. After this, a 1 mm sieve and tweezers were used to manually remove any visible pieces of rocks, wood, and other plant matter. Each sample was then placed in a ball mill for 5 minutes to form a fine powder which was placed in assimilation vials for laboratory work which tested for the quantities of Nitrate (NO_3), Ammonium (NH_4),

total organic Carbon (TOC) and Phosphorus. Nitrate, Ammonium and Phosphate (PO_4) were extracted from soil samples using KCl or Truog's solution, then the quantity of each compound was found using automated colorimetry. TOC was measured by flash combustion and an elemental analyzer. To compare the nutrient levels in the different burn plots, samples were grouped by burn plot and the data for each given plot was tested for normality. If the nutrients were found to follow a normal distribution, ANOVA testing was conducted in SPSS to compare nutrient levels between the burn plots. If the data did not follow a normal distribution, a nonparametric Kruskal Wallis test was conducted to the same end. Significant results would have been followed up by Tukey tests in order to which specific plots significantly differed with regards to nutrient levels.

Methods: Light availability

Finally, the light availability to the understory was approximated at the various plots using the abundance and mean DBH of adult trees as proxies inversely related to light availability; adult trees were defined as those with a DBH greater than 8 cm and only those within 3m of the transects were recorded. Canopy density was used to estimate the amount of light available in the understory as a tree with a larger DBH should result in a greater area of the canopy with being covered with foliage. The abundance and mean DBH of adult trees were recorded at each given transect, and regressions were conducted in SPSS to test both the relationship adult tree abundance and ground cover, as well as the relationship between adult tree mean DBH and ground cover.

Results

Results: Light availability

The mean and variance of the DBH of adult trees (DBH > 8 cm), adult tree abundance, and the mean and standard deviation of ground cover were calculated for each burn plot (Figure 1). This information was then used to conduct regressions in SPSS and test both the correlation between abundance of adult trees and ground cover and the correlation between adult tree mean DBH and ground cover. The results indicated that ground cover does not significantly correlate with either adult tree abundance or mean DBH (t = .581, p > .05, t = -.032, p > .05; Figure 2, Figure 3, Figure 4).

Results: Biodiversity

The average Shannon Index, species richness, equitability, and Simpson's Index were calculated for each burn plot (Figure 5). This information was used to conduct normality testing for both Simpson's Index and the Shannon Index; the distributions for both indices were found to be normal (d.f. = 4, p > .05; Figure 6). This enabled us to conduct ANOVA testing and test the null hypothesis that there would be no difference in biodiversity among the burn plots. These tests returned statistically significant results for both Simpson's Index and the Shannon Index, indicating that there was a real difference in biodiversity between the burn plots (F = 6.481, d.f. = 15, p < .05; F = 7.469, d.f. = 15, p < .05; Figure 7). Post-hoc Tukey tests were then conducted to find statistically significant differences between specific burn plots. The results of these tests indicated that the 2017 burn plot had a significantly lower Shannon Index and a significantly lower Simpson's Index (p < 0.05, p < 0.05; Figure 8).

Results: Soil

The average percentage of total organic carbon (%TOC) and amount of nitrates (NO3 ug / soil g), ammonium (NH4 ug / soil g), and phosphates (PO4 ug / soil g) were calculated for each burn plot (Figure 9). Normality tests were conducted for all of these variables in each burn plot;

results indicated that the data for %TOC and NH4 were normally distributed, whereas the data

for NO3 and PO4 were not (p > .05; p < .05; Figure 10). Accordingly, ANOVA was conducted

to compare the mean levels of TOC and NH4 among the burn plots. Results indicated that there

were no significant differences in the levels of TOC or NH4 between the burn plots (F = .662,

d.f. = 23, p > .05; F = .343, d.f. = 23, p > .05; Figure 11). Nonparametric Kruskal Wallis tests

were conducted to compare the mean levels of NO3 and PO4 amongst the burn plots. Results

indicated that the levels of neither NO3 nor PO4 differed significantly between the burn plots

($\square^2$ = .681, d.f. = 3, p > .05; $\square^2$ = 3.051, d.f. = 3, p > .05).

Figures

Figure 1: Adult tree mean DBH and abundance; mean and standard deviation of understory ground cover

Burn Plot	Mean DBH (cm)	Variance (cm)	Adult Tree Abundance	"Ground cover" (m)	Std dev of ground cover(m)
1911	16.866	13.313	29	9.170	4.621
1936	15.724	8.491	21	21.232	3.351
1980	10.554	2.175	36	21.510	2.813
2017	0	0	0	14.218	4.219

Figure 2: Regressions of DBH & Abundance against ground cover

Coefficients[a]

Model		Unstandardized Coefficients B	Unstandardized Coefficients Std. Error	Standardized Coefficients Beta	t	Sig.
1	(Constant)	28.635	9.841		2.910	.033
	DBH	.387	.667	.283	.581	.587
	ABUNDANCE	-.025	.779	-.015	-.032	.976

Figure 3: Scatter plot charting ground cover (m) against adult tree abundance
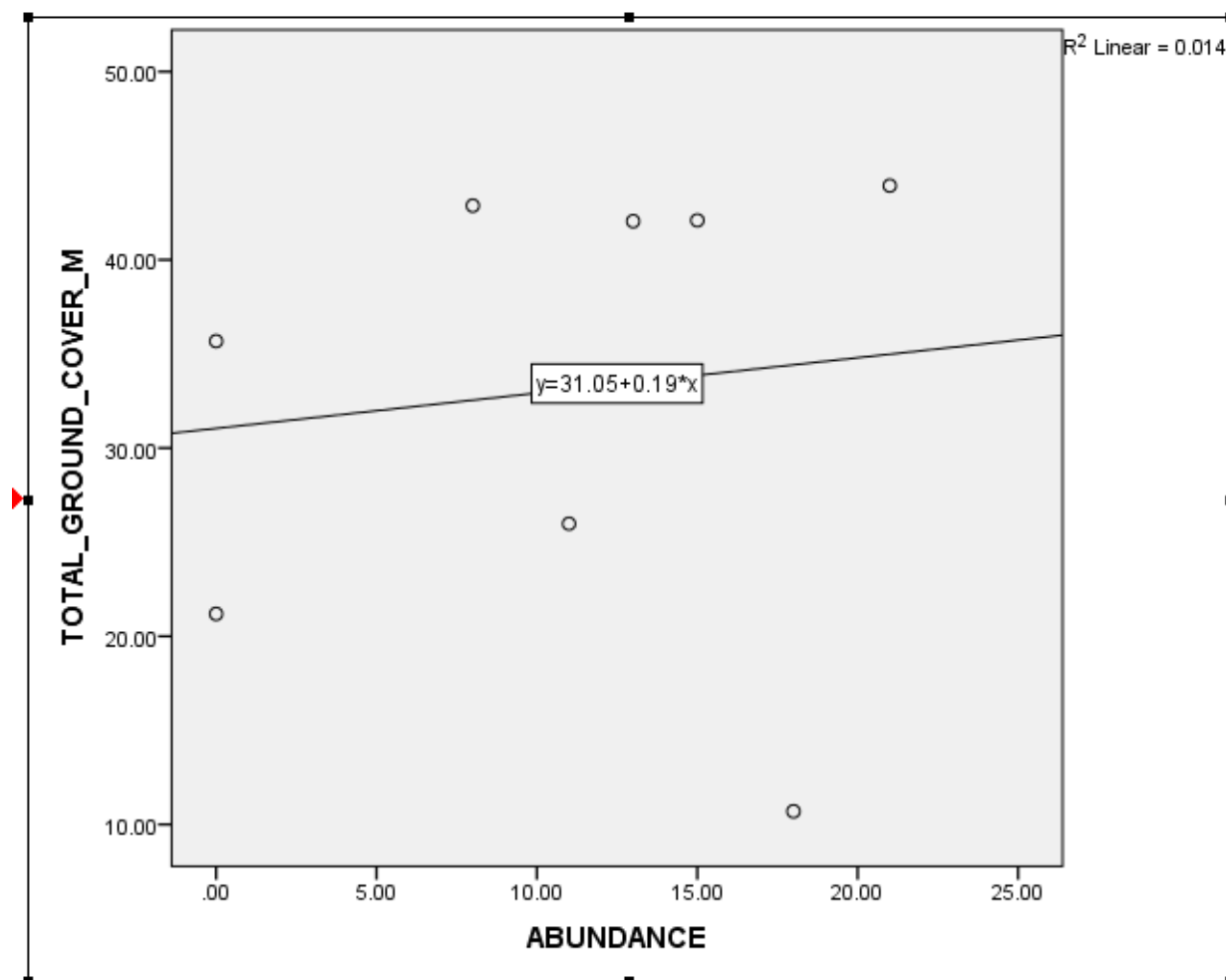
Figure 4: Scatter plot charting ground cover (m) against adult tree mean DBH

Graph

Figure 5: Average Shannon Index, Species Richness, Equitability, and Simpson's Index for each burn plot

Burn Plot	Shannon (H)	S	Equitability (E)	Simpson's (D)
1911	1.332	7	0.691	0.349
1936	1.402	8.25	0.674	0.374
1980	1.248	7.25	0.633	0.374
2017	0.641	5.75	0.384	0.687

Figure 6: Normality tests for Simpson's Index and Shannon Index in each burn plot

Tests of Normality

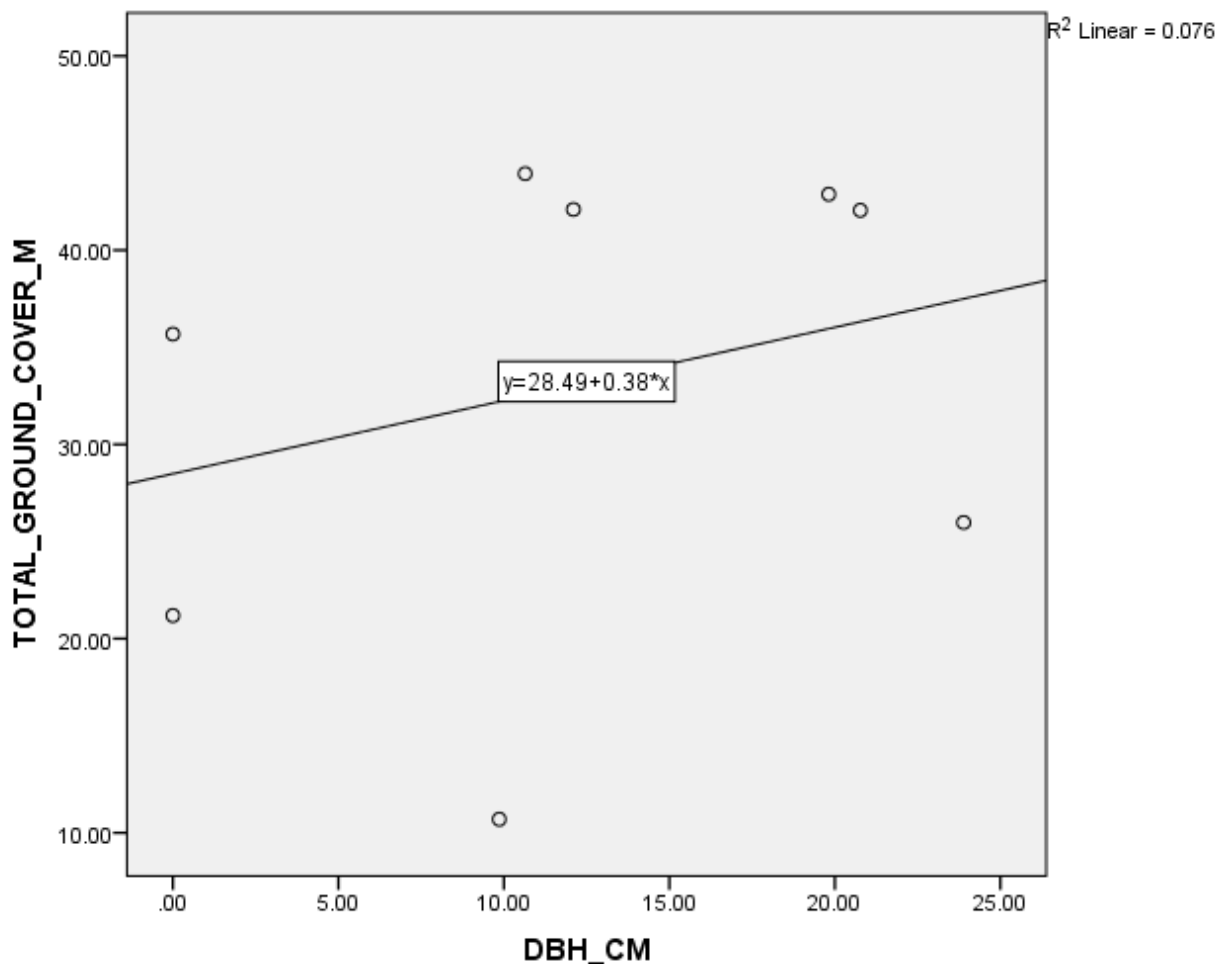|	BURN_PLOT	Kolmogorov-Smirnov[a]			Shapiro-Wilk		
		Statistic	df	Sig.	Statistic	df	Sig.
SIMPSON	1911.00	.277	4	.	.857	4	.251
	1936.00	.197	4	.	.985	4	.932
	1980.00	.277	4	.	.871	4	.303
	2017.00	.247	4	.	.925	4	.564
SHANNON	1911.00	.248	4	.	.957	4	.762
	1936.00	.302	4	.	.824	4	.153
	1980.00	.251	4	.	.860	4	.261
	2017.00	.259	4	.	.868	4	.291

Figure 7: ANOVA tests for Simpson's Index and Shannon Index

ANOVA

		Sum of Squares	df	Mean Square	F	Sig.
SIMPSON	Between Groups	.312	3	.104	6.481	.007
	Within Groups	.192	12	.016		
	Total	.504	15			
SHANNON	Between Groups	1.461	3	.487	7.469	.004
	Within Groups	.782	12	.065		
	Total	2.243	15			

Figure 8: Tukey's tests comparing Shannon and Simpson's Indices between burn plots

Post Hoc Tests

Multiple Comparisons

Tukey HSD

Dependent Variable	(I) BURN_PLOT	(J) BURN_PLOT	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
SHANNON	1911.00	1936.00	-.07012	.18055	.979	-.6062	.4659
		1980.00	.08411	.18055	.965	-.4519	.6202
		2017.00	.69100*	.18055	.011	.1550	1.2270
	1936.00	1911.00	.07012	.18055	.979	-.4659	.6062
		1980.00	.15422	.18055	.828	-.3818	.6903
		2017.00	.76112*	.18055	.006	.2251	1.2972
	1980.00	1911.00	-.08411	.18055	.965	-.6202	.4519
		1936.00	-.15422	.18055	.828	-.6903	.3818
		2017.00	.60689*	.18055	.025	.0708	1.1429
	2017.00	1911.00	-.69100*	.18055	.011	-1.2270	-.1550
		1936.00	-.76112*	.18055	.006	-1.2972	-.2251
		1980.00	-.60689*	.18055	.025	-1.1429	-.0708
SIMPSON	1911.00	1936.00	-.02487	.08952	.992	-.2906	.2409
		1980.00	-.02564	.08952	.991	-.2914	.2401
		2017.00	-.33825*	.08952	.012	-.6040	-.0725
	1936.00	1911.00	.02487	.08952	.992	-.2409	.2906
		1980.00	-.00077	.08952	1.000	-.2665	.2650
		2017.00	-.31338*	.08952	.020	-.5791	-.0476
	1980.00	1911.00	.02564	.08952	.991	-.2401	.2914
		1936.00	.00077	.08952	1.000	-.2650	.2665
		2017.00	-.31261*	.08952	.020	-.5784	-.0468
	2017.00	1911.00	.33825*	.08952	.012	.0725	.6040
		1936.00	.31338*	.08952	.020	.0476	.5791
		1980.00	.31261*	.08952	.020	.0468	.5784

*. The mean difference is significant at the 0.05 level.

Figure 9: Concentrations of TOC, Nitrates, Ammonium, and Phosphate among burn plots

Burn Plot	%TOC	NO3 (ug/g)	NH4 (ug/g)	PO4 (ug/g)
1911	11.455	27.06833333	1047.36	0.02666666667
1936	11.57333333	10.435	924.015	0.045
1980	7.066666667	18.69	799.1516667	0.03666666667
2017	14.28333333	21.868	1203.191667	0.04

Figure 10: Normality tests for TOC, NO3, NH4, and PO4 for each burn plot

Tests of Normality

	BURN_PLOT	Kolmogorov-Smirnov[a]			Shapiro-Wilk		
		Statistic	df	Sig.	Statistic	df	Sig.
TOC	1911.00	.186	6	.200[*]	.944	6	.688
	1936.00	.235	4	.	.922	4	.547
	1980.00	.209	5	.200[*]	.921	5	.533
	2017.00	.301	5	.159	.788	5	.064
NO3	1911.00	.295	6	.112	.781	6	.040
	1936.00	.253	4	.	.891	4	.387
	1980.00	.359	5	.034	.768	5	.043
	2017.00	.205	5	.200[*]	.878	5	.299
NH4	1911.00	.209	6	.200[*]	.892	6	.330
	1936.00	.258	4	.	.886	4	.367
	1980.00	.258	5	.200[*]	.888	5	.348
	2017.00	.239	5	.200[*]	.901	5	.418
PO4	1911.00	.293	6	.118	.777	6	.036
	1936.00	.252	4	.	.916	4	.513
	1980.00	.180	5	.200[*]	.952	5	.754
	2017.00	.430	5	.003	.596	5	.001

Figure 11: ANOVA tests for TOC and NH4

ANOVA

		Sum of Squares	df	Mean Square	F	Sig.
TOC	Between Groups	160.508	3	53.503	.662	.585
	Within Groups	1617.221	20	80.861		
	Total	1777.729	23			
NH4	Between Groups	536825.488	3	178941.830	.343	.794
	Within Groups	10424921.61	20	521246.080		
	Total	10961747.09	23			

Figure 12: Kruskal Wallis test for NO3 and PO4

Test Statistics[a,b]

	NO3	PO4
Chi-Square	.681	3.051
df	3	3
Asymp. Sig.	.878	.384

a. Kruskal Wallis Test

b. Grouping Variable: BURN_PLOT

Discussion

According to the data, there is a statistically significant difference in both the species richness and evenness (Shannon-Wiener and Simpson biodiversity indices, respectively) between the 2017 plot and the other three plots. However, there is no statistically significant difference in either of the biodiversity parameters between the other three plots. As a result, we reject the null hypothesis that there is no difference in biodiversity between the burn plots. These results are likely due to the fact that before the 2017 burn, the most recent plot studied was burned in 1980, which was 38 years before sampling. Data from previous studies has shown a similar pattern; biodiversity increases for approximately 40 years following a disturbance, coinciding with the recolonization of species, and then remains fairly constant (Moorhead 1996). Thus, the data on biodiversity supports this timeline of secondary succession stating that the majority of successional changes occur by forty years post-burn. In terms of biodiversity restoration of burned areas, this indicates that the first forty years following a burn are the most significant, and comprise the time period in which most interventions should be taken in terms of controlling what species recolonize and at what rate.

The data also shows no statistically significant difference in any of the soils tested between the sampled plots and therefore we fail to reject the null hypothesis that there will be no difference in the nutrient levels between the burn plots. This may be due to soil variance within each plot itself, perhaps due to uneven burning or by heterogeneity within the plots in terms of both species composition and density, which could impact the composition of the A-horizon that was sampled (Finzi et. al. 1998). Furthermore, it is possible that the use of fairly constant sampling depth led to incidental sampling of the O-horizon or B-horizons of the sampled soils, due to uneven depths of the A-horizon between the different plots, since the A-horizon is

typically converted to ash during burning, and slowly rebuilt with recolonization (Johnson et. al. 2001). Incidental sampling of anything other than the A-horizon may have significantly impacted the results, as levels of the tested nutrients, carbon and nitrogen in particular, vary significantly by soil horizon (Trumbore 1996).

Linear regression demonstrated that ground cover had no significant correlation with either adult tree abundance or mean DBH of adult trees, each of which were used as inverse proxies for light availability. One explanation for this lack of significant results includes the possibility that mean abundance and DBH of adult trees are ineffective proxies for approximating light availability. Additionally, while it is possible that understory ground cover decreases with decreased light availability, it is also possible that a shift to more shade-tolerant understory species could occur while maintaining fairly constant levels of ground cover (Canham 1988. Thus, it is also possible that there was a shift in understory species composition with decreasing light availability, instead of an overall decrease in ground cover. Generally, there is an anticipated increase in abundance of herbaceous vegetation early in the successional process, such as *Pteridium aquilinum*, followed by shade-intolerant woody plants such as *Populus grandidentata* and finally shade tolerant woody plants like *Pinus strobus* and *Acer rubrum* (Moorhead 1996).

There are a number of both noisy and confounding variables which should be considered as possible sources of error in the data. One prominent confounding variable is that while the number of adult trees was used as a proxy for light availability, juvenile trees (<8 cm DBH) surrounding the transects also block light (Bellow 2003). There are also a number of possible noisy variables which may constitute sources of error in the data, including geographic differences between the burn plots, differences in temperatures and evenness of the initial burns,

and climatic differences in the years since the initial burns. Herbivory of the understory and of adult trees could impact understory ground cover and composition, as well as light cover (Safford et al. 2006). Another possible source of error in the data is the way in which understory ground cover was calculated; by double-counting areas on which species overlapped on the sampling transect, this method rejects the more common convention of using percent cover without accounting for overlap of species (Hansen et al. 2003, Zhu et al. 1994) . This was done in an attempt to more effectively account for differences in levels of understory mass, but it is possible that it presents inaccurate data, especially because it did not account for species abundance via number of individuals.

It is similarly possible that errors could have arisen from non-random placement of transects. As locations were being chosen to lay transects, some were favored over others due to accessibility or other forms of bias. Temporal gaps in data collection could have caused errors as well. The temporal gaps are particularly relevant to the data on ground cover is because *Pteridium aquilinum*, which grow very quickly (Alonso-Amelot 1996), comprised a significant portion of the understory species. Since the first and last days of understory ground cover sampling fell over a week apart, this fast growth may have led to false recording of the later-sampled plots as having more ground cover.

The most obvious evidence of uneven burning could be seen in the most recent (2017) burn plot, by the uneven scorching of the ground and the presence of standing juvenile trees which were clearly established before 2017. Furthermore, the large standard deviation of the length cover between the tested transect segments relative to the 1936 and 1980 burns (see Figure 1) is likely due to the highly uneven distribution of understory ground cover due to the uneven burning (Kennedy 2006). The most burned areas had visible ash as most or all of the A-

horizon had burned and converted to ash; these areas had few or no understory inhabitants. The earlier burns may also have been burned unevenly, but obvious evidence of this on the older burn plots is not likely to be present. The presence of uneven or patchy burning has the potential to impact all aspects of the data that was collected. This is because the presence of burning in a given area is necessary to catalyze the process of secondary succession, and the degree of initial burning has the potential to impact the long-term results of this process (Moorhead 1996) which would include long-term impacts on things like soil composition and understory biodiversity.

By gaining a deeper understanding of the successional processes which occur after a major burn in a temperate hardwood forest, more effective methods may become available for the management of these ecosystems after a burn. This is particularly relevant because most existing data on secondary succession was collected in ecosystems in which naturally-occurring fires are currently more common (Hanes 1971, Keeley 1981, Adler et al. 1998), and naturally-occurring forest fires are likely to become increasingly common in temperate ecosystems with ongoing climate change (Handler et. al 2014). Thus, any differences in the effective management of burned temperate ecosystems in particular should be preemptively explored, before these fires become more common and management is immediately necessary.

A better understanding of secondary succession in temperate hardwood forests may help to better contextualize the longevity of biodiversity loss following a major burn in this type of ecosystem, thus allowing for more appropriate future allocation of resources to wildfire prevention and recovery. Additionally, a more thorough understanding of the long-term changes in biodiversity following burns may be used to inform public policy in regards to both fire prevention and the accessibility of controlled burns as a tool in ecological management. Furthermore, the analysis of various abiotic factors, such as light and nutrient availability, and

how they vary based on amount of time post-burn, allow for a better understanding of how these important abiotic factors are impacted, and at what rate, through the process of secondary succession. This may allow for more effective management of burned ecosystems via manipulation of these abiotic factors. Finally, understanding the ways in which controlled burns can impact the short-term and long-term biodiversity of a forest understory, and understanding the possible longevity of these changes, is an integral part of sustainable forestry (Lindenmayer 2000). Since the preservation of biological diversity is already one of the most important goals in sustainable ecological management (Lindenmayer 2000), this understanding will only become more important as naturally-occurring fires become a more realized threat in regions further and further north in accordance with climate change (Handler et al. 2014).

There are a number of ways in which this experiment could be improved with future replication. Likely the most important improvement is to incorporate more recent burn plots; it is established that most of the successional changes occur in the first forty years following a burn (Moorhead 1996) and the use here of the 1980, 1936, and control plot means that three out of four plots were approximately this old, or older at time of sampling. This is likely a major contributor to the lack of significant data showing differences in biodiversity between the three older plots in particular. This timeline is supported by the data, which shows that species evenness and richness both vary between the 2017 burn and all of the others, but they do not vary significantly from each other. If further replication is done on the UMBS burn plots, continued use of the 2017 burn is recommended, along with use of the nearby 1998 burn, which would allow for some insight into some of the earlier successional changes that occur following a major burn in this ecosystem. Since the majority of successional changes are likely to occur within forty years since burning (Moorhead 1996), a better understanding of the rate of

biodiversity restoration within this window will be integral in the management of burned areas immediately following a major burn.

It is also recommended that future attempts to evaluate the soil composition of the burn plots explore the impacts of uneven burning on soil composition on a recent plot where uneven burning is evident. By comparing composition data from more-burned and less-burned areas of the same plot, it may be possible to better quantify the impacts of burning on soil composition, and of uneven burning on soil data collected from the plots as a whole. Additionally, if another attempt to evaluate the A-horizon of the plot soils is made, a consistent method of evaluating the depth of the A-horizon should be established. With different layers of ash on the more recent burns, and different depths of the A-horizon among the plots due to the burning of the A-horizon, it may otherwise be difficult to decide to what depth the soil should be sampled. It is possible that use of a soil corer could help with consistent sampling of only the A-horizon.

While understory abundance was approximated via amount of ground cover, future studies might try to incorporate an element of abundance into their evaluation of the understory as well, perhaps by evaluating ground cover in addition to counting the number of individuals present along the sampling transect. Including data on abundance may provide more accurate criteria to calculate biodiversity, in which proportions of different species are vital. Lastly, it is recommended that for future studies which attempt to evaluate the impacts of light availability on understory ground cover or biomass account for differences in the shade tolerance of different understory species, as a transition toward more shade-tolerant species is likely through the process of secondary succession (Moorhead 1996), and this may prevent reductions in ground cover or abundance even with decreased light availability.

Acknowledgements

We would like to thank everyone at University of Michigan Biological Station for their support. We also want to give a special thanks to Timothy Veverica and the chemistry lab staff for analyzing the soil samples and Adam Schubel for sharing a wealth of information with us regarding the burn plots and their inhabitants. We would also like to thank Lucas Nave for his input on our study design. Lastly, a special thanks to our instructurer Jasmine Crumsey Forde for her help with editing our paper and with our project as a whole, and to our TAs Corbin Kuntze and Olivia Brinks for helping us edit both our presentation and paper.

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