

FUEL MANAGEMENT CONSIDERATIONS IN CARIBBEAN PINE
FORESTS OF THE MOUNTAIN PINE RIDGE OF BELIZE, CENTRAL
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FUEL MANAGEMENT CONSIDERATIONS IN CARIBBEAN PINE FORESTS OF
THE MOUNTAIN PINE RIDGE OF BELIZE, CENTRAL AMERICA

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Dedicated to Janet Lynn Colatskie (Mom) 1954-2000

You left us early, but your imprint will persist throughout my life

Abstract

Situated within the Maya Mountains of Central America, the Mountain Pine Ridge (MPR) ecosystem encompasses approximately 76,000ha (107,000 acres) within the country of Belize. Dominated by Caribbean pine (*Pinus caribaea*), the MPR is a fire-dependent ecosystem that is currently in transition following a devastating 2001 outbreak of the Southern pine beetle (*Dendroctonus frontalis*). The beetle outbreak was preceded by an era of fire suppression which recruited dense stands of pine with a hardwood understory. Currently the ecosystem is a patchwork of varying forest conditions from open savanna to stands of pine exceeding 7,000 trees per acre (2832 trees per hectare). Threatening the pine regeneration is the buildup of fuels such as dense thickets of tiger fern (*Dicranopteris pectinata*), which can carry intense fires. In early 2010, thirty nine plots were established within the eastern portion of the MPR to determine the variation in abundance and height of a variety of fuels including major physiognomic plant groups, coarse woody debris, bare mineral soil, and percent cover of litter within designated units of varying times since fire. The fire return interval units were grouped into three separate fire intervals including recent (1-3 years since fire), midrange (4-10 years) and extended (11+ years). Plant physiognomic groups were also evaluated in plots of varying stand densities and canopy coverage. As expected, hardwoods, shrubs, and litter significantly increase in abundance in extended fire intervals. As expected grasses showed a negative correlation with canopy coverage while tiger fern showed a positive correlation. With respect to tiger fern abundance, GIS analysis was conducted to determine if aspect had an effect, however no significant differences were found.

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Introduction

Situated within the Maya Mountains, the Mountain Pine Ridge (MPR) Ecosystem encompasses approximately 76,000 hectares (192,000 acres) in the west-central portion of the Central American country of Belize. The MPR Forest Reserve, owned by the Belize government, encompasses 44,000 hectares (107,000 acres), comprising the largest landholding in the MPR. The remaining lands in the MPR ecosystem are owned by private landowners.

The MPR ecosystem is a fire-dependent community dominated by Caribbean pine (*Pinus caribaea*) with a variable component of Honduran pine (*Pinus oocarpa*). The MPR is currently in transition following the devastating 2001 outbreak of the southern pine beetle (SPB) (*Dendroctonus frontalis*). The outbreak caused an estimated 70-90 percent mortality of mature pine, thereby, setting back the forest to an early stage of succession (Headley 2008, FAO 2008). The beetle outbreak was preceded by an era of intense management which included logging and fire suppression beginning in the 1940s in an effort to protect pine regeneration from fires that commonly burn during the dry season which extends January through June. Currently, the MPR is a patchwork of pine regeneration from dense regeneration > 2,820 trees per ha (>7,000 trees per acre) to open savannahs invaded by tiger fern (*Dicranopteris pectinata*) in some areas.

Background

Cultural and Historical Perspective

Belize, with an area of 22,966km² (8,966mi²), is located on the Yucatan peninsula and is bordered by Mexico to the North and by Guatemala to the West and South (Figure 1). Formerly a British colony known as British Honduras, Belize gained its independence in 1981. The British had officially held Belize following a treaty signed with Mexico in 1897; however, Europeans had maintained a presence in the region since the 17th century. As was the fate of many European colonial assets, Belize's natural resources suffered heavily from extraction. However prior to the time Europeans settled in the region, human presence had been firmly established with the Mayan culture as early as 1500 BC, with initial hunter-gatherer cultures existing centuries prior.

In terms of timber, valuable hardwood species of the hardwood rainforest, such as Mahogany (*Swietenia macrophylla*), were the initial target of loggers. During the early 20th century, logging activities included other regions, such as the MPR, where cypress (*Podocarpus guatemalensis*) and pine (*Pinus caribaea*) were harvested.

Culturally, Belize is unique in that it is the only Central American country to carry a British colonial heritage. The low population density of Belize at 14.6/km² (38/mi²) is also unique relative to surrounding countries such as Guatemala, which has a population density of 128/km² (334/mi²). This low population density offers potential in terms of conservation of biodiversity in this small country. The growth rate of the region is relatively high compared to neighboring countries, with a growth rate of 2.056% and a present population of 310,000 with 36% of the population under 14 years of age (CIA,

2011). Racially, Belize is a melting pot of cultures with the principal groups including Mestizo, Creole, Mayan, Asian, Mennonite, European, and American cultures.

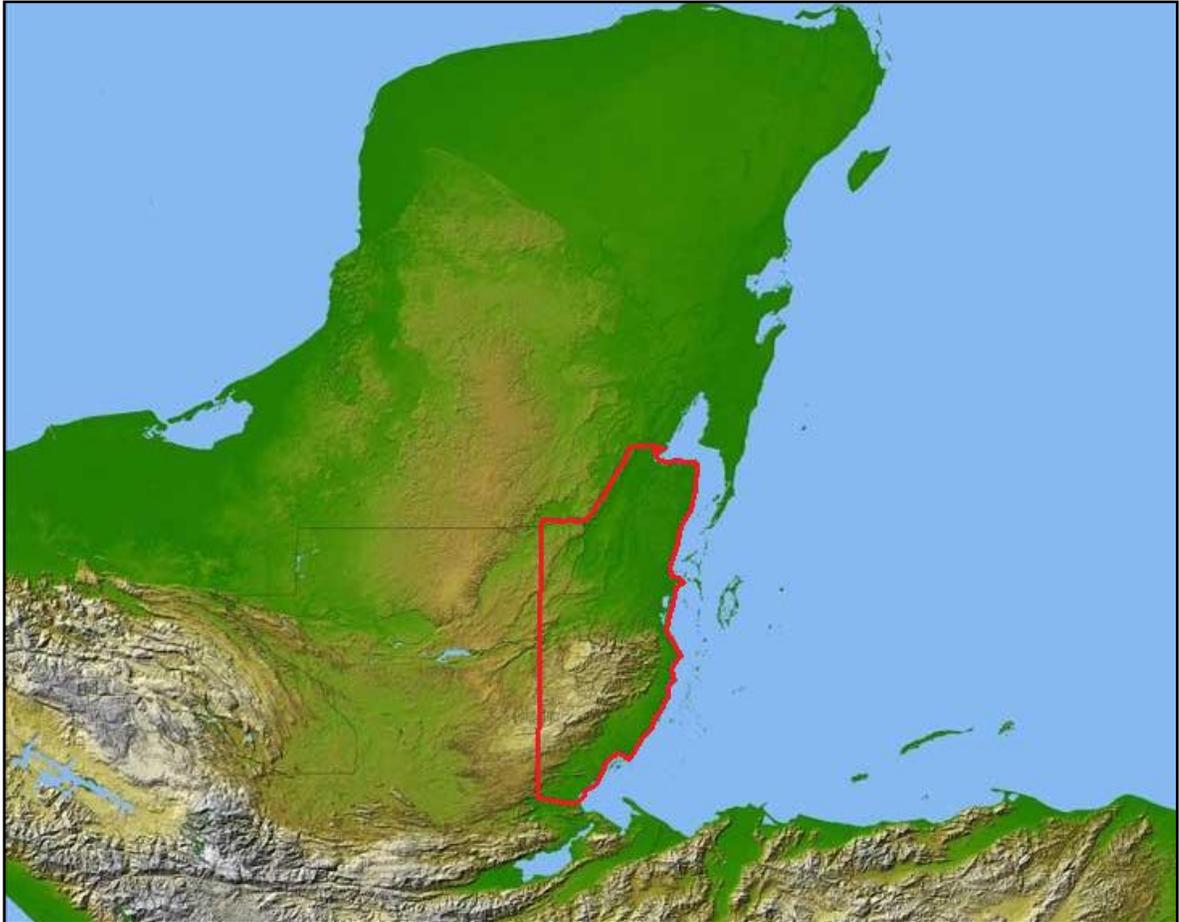


Figure 1. Map of Belize (Outlined in red). Belize is bordered to the West by Guatemala and to the North by Mexico. Image courtesy of NASA (http://visibleearth.nasa.gov/view_rec.php?id=9199).

The Mountain Pine ‘Ridge’

The MPR is situated on a granitic dome that encompasses a hilly plateau ranging from 400 to 1,000 meters above sea level within the west-central portion of Belize (Johnson and Chaffey 1973).

The MPR supports a variety of habitat types including upland pine-oak savannah, broadleaf forests, palm bogs, a variety of streams, as well as varying degrees of agricultural uses. The most abundant habitat type of the entire MPR region is the pine-oak savannah/pine-oak woodlands or 'pine ridge' by which it is referred to locally (the term 'ridge' refers to vegetation types and is not to be confused with topography), this vegetation type once comprised 75-80% of the MPR (Means 1996). The distribution of savannahs were confined to the non-calcareous parts of the reserve and consisted of a generally low stocking of young and naturally-regenerated pine with an open understory of small trees, shrubs and a varying mixture of ground vegetation (Johnson and Chaffey 1973).

The MPR is a fire-dependent ecosystem that historically persisted in a savanna condition with a sparse canopy dominated by interspersed Caribbean pine (*Pinus caribaea*), Honduran pine (*Pinus oocarpa*) and oaks (*Quercus* spp). Pither and Kellman (2001) point out that charcoal and pollen evidence indicate that the savanna condition of the ecosystem had been in existence for at least 11,000 years and was possibly a remnant of a much larger system of savannas that were present throughout Central America during the last glacial maximum approximately 20,000 years ago. As is the case with most savannah ecosystems, the richest species diversity of the MPR resides within the ground layer vegetation. A diverse array of grasses, ferns, sedges, and shrubs populate the ground layer and during the dry season the vegetation will desiccate and carry fires through the forest.

The persistence of the savannah was due to frequent fires and poor soil fertility which precluded the establishment of a closed canopy forest. Without frequent fire, an essential

component of the ecosystem, the open savanna condition will give way to losses in understory vegetative diversity and highly stocked stands of pine mixed with an understory of hardwoods (Myers, O'Brien, & Morrison, 2006).

Climate

At 17° N latitude Belize experiences a humid, subtropical climate, yet this varies depending on location and altitude. For example, annual rainfall ranges from 127cm (50") in northwestern Belize to 450cm (177") recorded in the southern portion of Belize. A general north to south gradient exists with considerably more rainfall occurring in the southern portion of the country (Penn, Sutton, & Monro 2003). Even within the MPR, annual rainfall totals can be quite variable. The majority of the MPR resides in the rain shadow of the Maya Mountain massif (Johnson & Chaffey, 1973). Typically, moisture is brought in on trade winds from the Caribbean Sea, rising over the eastern portion of the escarpment. Orographic lift causes a gradient of moisture within the MPR with more precipitation falling on the eastern portions of the pine ridge with less precipitation towards the west.

The climate pattern is variable from year-to-year but generally the MPR experiences a dry season of limited precipitation and warm weather lasting from February until May (Figure 2). Temperatures vary from 9° C (48° F) in January to over 38°C (100°F) during the peak of the dry season in May. Unlike temperate climates with peak temperatures in the mid-summer, the MPR experiences peak temperatures during April and May (Figure 2). The 'dry' season is followed by sporadic storms during the summer months with heavier long-lasting rainfall events occurring in November and December. The

combination of high temperatures, low humidity, low rainfall, and high incidence of electrical storms is highly conducive to the ignition of natural fires by lightning as well as ignition by anthropogenic sources, both accidental and induced, during the dry season (Means 1997).

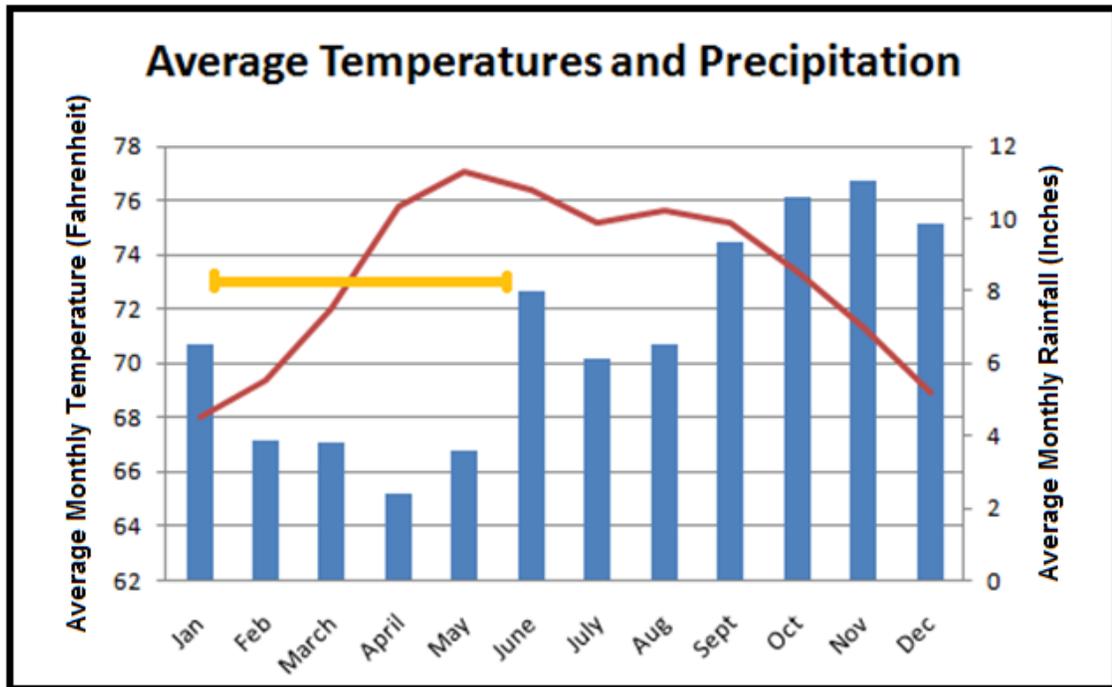


Figure 2. Average monthly temperatures (F°) (red line) and average monthly precipitation (inches)(blue bars) recorded from 1983-1996. Data recorded at the Hidden Valley Inn in the Eastern portion of the MPR. Recorded by John McLaughlin and presented in Means (1997). The yellow bar indicates the average extent of the dry season.

Geology

The MPR ecosystem owes its existence to the underlying geological features. Means (1997) describes Belize and adjacent parts of Mexico and Guatemala as a vast limestone platform with a large “island” of much older rocks protruding from it in the form of the Maya Mountains. The MPR is part of an uplifted fault block composed of metamorphic,

igneous, and sedimentary rocks dating from the Pennsylvanian to the early Cretaceous periods (Dubbin, Penn, & Hodsona, 2006). During the Permian period (280-300 million years ago) the area consisted of folded metasediments, a group of rocks which fall into the Santa Rosa group (Ennion, 1995). Over time, granite intruded the Santa Rosa stones during the Triassic period (220-230 million years ago). During the Cretaceous period, (140-180 mya) the area was submerged under a shallow sea and layers of limestone were deposited over the Santa Rosa metasediments as well as the protruding granites. It is suggested the Bald Hills, which persist at higher elevations of greater than 2,700 feet (700m), were not submerged and subjected to continued weathering. Following the recession of the shallow seas the limestone veneer weathered except for scattered areas within the western portion of the MPR where limestone capped hills still persist. Presently the MPR is a highly dissected region with exposed granite, Santa Rosa metasediments, and surrounded by limestone hardwood dominated forests.

Soils

As within all forest ecosystems, species abundance and diversity are constrained by a number of factors including nutrient deficiencies and water-holding capacity limitations. In the MPR, these limitations are imposed by the coarse-textured and highly-weathered ultisols derived from metasediments and granitic parent materials (Figure 3) (Dubbin, Penn, & Hodsona, 2006). The soils of the MPR fall within one of two suites. The Stopper suite is derived from granitic parent materials and the Ossory suite is derived from metasediments. The Stopper suite includes the pinol subsuite while the Ossory subsuites include Cooma, Baldy and Chiquibul (Ennion, 1995).

The pinol subsuite soils are formed from granitic materials and generally weather into coarse loamy sand to clay loam topsoil with a sandy loam-clay subsoil. This soil is characterized the presence of angular quartz materials. Being born of granitic parent materials these soils are the most resistant to weathering than those of the metasedimentary Ossory subsuites. Wright et al., (1959) describe this soil as 'senile' in terms of soil development and found them to be highly acidic and lacking in base saturation. Johnson and Chaffey (1973) describes this subsuite as mature soils in the development sequence from yellow latosolic to red-yellow podzolic, formed on lime-poor parent material. Caribbean pine dominates on this soil, however, the abundance of pine is largely determined by fire and logging histories (Ennion, 1995).

On the eastern portion of the MPR the 'bald hills' persist. At 3,000 feet (700m) the bald hills occupy the highest topography in the MPR. The soils of the bald hills are classified under the baldy subsuite and are formed by weathering of underlying argillites, which are lithified clays. These soils are very shallow with stony loam topsoil. In addition, they are limiting to plant growth in that they are highly leached, acidic and base deficient and contain little phosphate (Ennion, 1995). The vegetation occurring on this soil type includes principally grasses and sedges with a few stunted Caribbean pine.

Also derived from metasediments are the deeper and more developed cooma subsuite soils. The topsoils of this subsuite are silty or fine sandy loam-clay loam with an underlying stony horizon which is above a silt clay layer (Ennion, 1995). Cooma soils are developed from Santa Rosa slates and have redder, siltier subsoils and contain less coarse sand relative to soils developed from granitic parent materials such as the pinol subsuite soils (Johnson & Chaffey, 1973). These soils are also nutrient poor with low available

phosphate. *P. caribaea* dominates on this soil with *P. oocarpa* occurring and dominating on the steeper slopes. When these soils are disturbed by anthropogenic practices such as logging they are often prone to invasion by tiger fern (*Dicranopteris pectinata*) or dumb cane (*Tripsacum latifolium*) (Kemp, 1972).

Occurring intermixed with and typically downslope from the soils of the Cooma subsuite are the Chiquibul subsuite soils. These soils occur on steeper slopes than the Cooma subsuite soils and are more prone to erosion. As they occur on steeper slopes they are shallower and less developed than the Cooma soils which make them droughtier than the Cooma soils upslope. The texture of chiquibul soils are typically sandy loam to clay loam (Ennion, 1995). Tiger fern may also invade these slopes when they are disturbed due to their predisposition to erosion. Caribbean pine may occur in these soils however on steeper slopes *P. oocarpa* dominate.

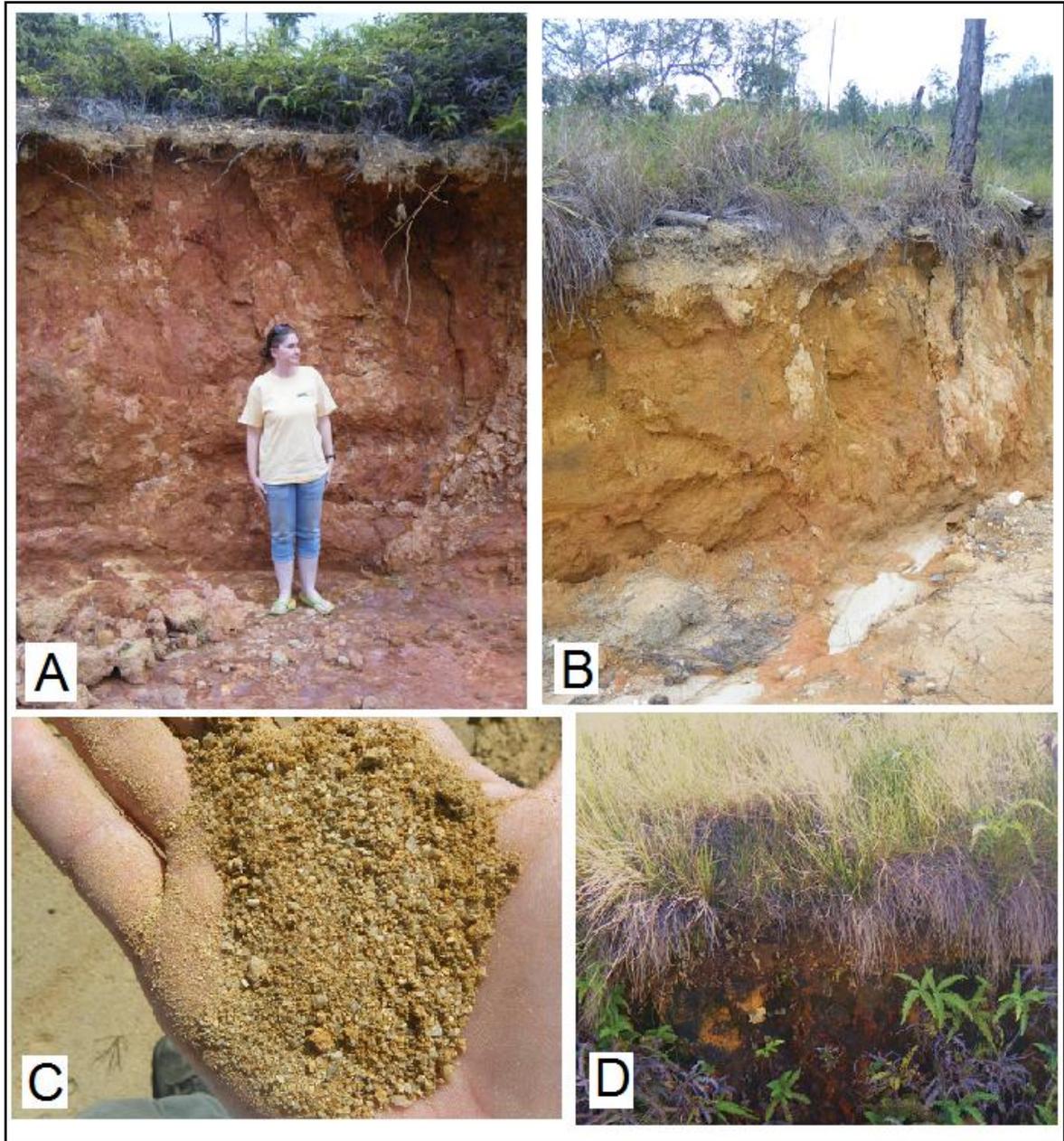


Figure 3. Soils of the Mountain Pine Ridge. The soils key out to Ultisols as they are highly weathered, acidic, nutrient poor soils with minimal water holding capacity. Soils developed from Santa Rosa slates, such as the cooma and chiquibul subsuites (Photo A) have red silty clay subsoils. Soils in the pinol subsuite are developed from granite and often contain coarse sands (Photos B and C). Soils of the baldy subsuite are very thin and stony due to intense weathering above argillites (Photo D).

Current Forest Ecosystem Conditions within the Mountain Pine Ridge

Penn et al. (2003) state that the MPR ecosystem is a dynamic assemblage of successional vegetation maintained in a subclimax state mainly by fire, hurricane, and more recently insect predation. Ecologists, including Means (1997), Penn et al. (2003) Meerman & Sabido (2001), and Pither and Kellman (2002), have attempted to delineate the variety of ecosystems that occur within the MPR. However, the effect of fire regimes on these ecosystems over time was not assessed prior to this study.

Means (1997) delineated fifteen distinct ecosystems including pine savanna, slope transition forest, exposed rocky cliff, broadleaf forest, hillside seepage bog, palm bog, floodplains, cascade rock face, plateau stream ravine stream, canyon stream, and a variety of ruderal habitats. Penn et al. (2002) characterized 32 specific vegetation classes within the greater Maya Mountains. Furthermore, Wright et al., (1959) classified MPR vegetation into three vegetation types including oak-pine-florazul forest, oak-pine-*Clusia* spp. forest with abundant silver palmetto, and savanna grass with scattered pine, oak, *Leucothos* spp. *Clusia* spp. and crabboo.

For the purpose of discussion within this publication, ecosystems within a forested setting can be broadly delineated within one of three categories including: pine-oak savanna/pine-oak forest, pine/broadleaf transition forest, and broadleaf riparian forest (Figure 5).

Pine-Oak Savannah and Pine-Oak Forest

Historically, the pine-oak savannah/pine-oak forest was the most abundant forest type within the Mountain Pine Ridge, once covering 75-80% of the MPR (Means, 1997). The persistence and the reason for the origin of the pine savannah through time is debatable. Hutchinson (1976) discusses two theories for its origin; one in which the acidic soils were the main driver for the existence of the MPR, the second theory holds that anthropogenic disturbance through agricultural use and fires produced acidic soil which was populated by the pine-oak savannah ecosystem. Hutchinson (1976) contends that the soils were poor to begin with and that native cultures may have expanded the range of the savannah ecosystem and increased the frequency of fires. The most refutable reason for the anthropogenic creation hypothesis is that palynological (pollen) data suggest the savannah ecosystems were at their most extensive 1000-4000 BP and prior to the peak of the Maya civilization (Wyatt, 2008).

The pine-oak savannah ecosystem is a globally unique tropical resource. In the publication regarding tropical American savannas, Beard (1953) defined a savannah as

“a plant formation of tropical American comprising a virtually continuous, ecologically dominant stratum of more or less xeromorphic herbs, of which grasses and sedges are the principal components, which scattered shrubs, trees or palms sometimes present, the essential point is that the herb stratum is ecologically dominant”.

With its historic ground vegetation layer rich in grasses and sedges, shrubs, with scattered oaks and pines, the MPR as a whole once fit Beard’s description. Means (1997)

describes the historic MPR savannah as a region of sparse pine (*P. caribaea*) with scattered oaks (*Quercus* spp) and a rich diversity of hardwoods, shrubs, grasses/sedges, and forbs (Figure 5.C). Following the fire suppression efforts of the 1940's however, the biomass of the ground vegetation was lost as they were shaded out by dense stands of pine and hardwoods which were once thinned by periodic fires. Following the reintroduction of fire through prescribed fire, this ecosystem shows potential for restoration to its historic state. Depending on a variety of factors including fire, silvicultural use, and insect predation, current stands of pine can be found in a variety of densities. These densities can be categorized into four broad groups relative to silvicultural needs (Figure 4).



Figure 4. Four typical pine stand conditions of the MPR. Upper left is an appropriately thinned stand, upper right is overstocked, lower left is lacking in regeneration and lower right shows mortality from an intense wildfire.

Broadleaf/Riparian Forest

Paralleling the streams which traverse the MPR is what is commonly referred to as ‘hardwood stringers’ or broadleaf/riparian forests (Figure 5.B) The richer soils and higher humidity, due to greater canopy structure and closure, support a variety of hardwoods at the foot of these slopes which can also be found in the broadleaf lowlands surrounding the MPR. These islands or fragments of ancient broadleaf forests are unique in that they persist within a pine savannah forest allowing for observation of long-term fragmentation from larger tracts of broadleaf forests (Pither & Kellman, 2002). Pither and Kellman (2002) further demonstrate the uniqueness of these communities as they found 49.2% of the species were exclusively humid forest species, 27.8% occur in both rainforest and other communities, and 13.9 % are not typically found in broadleaf rainforests. Species occurring in this ecosystem include mahogany (*Swietenia macrophylla*), santa maria (*Calophyllum brasiliense*), and nargusta (*Terminalia amazonia*), among others (Means, 1997). Pither and Kellman (1997) identified 144 species of hardwoods which comprises 1/5th of the 700 native tree species of Belize. The presence of this ecosystem augments the biodiversity of the MPR. As evidence indicates the MPR savannah has been in existence for at least 11,000 years, it is likely that the riparian broadleaf forests have been in existence for the same period of time (Meave & Kellman, 1994).

With their dense canopy, moist litter layer, and proximity to perennial streams, these sites serve as naturally occurring firebreaks. However, during extreme fire weather conditions fuels in these regions may be dry enough for ignition allowing for fires to move through, particularly where these bands of broadleaf forests are narrow. Kellman and Meave (1997) indicate the uncommon fires in the broadleaf/riparian forests have had an

enrichment in terms of biodiversity instead of a negative effect on the forest, which provide refugia for broadleaf forest species in the fire-prone region of the MPR. The disturbance of fire creates a heterogeneous environment allowing for a gradation of both hardwood and savannah species.

Pine/Broadleaf Transition Forest

Straddling sites between the pine savannah/forest and the broadleaf forest is the pine/broadleaf transitional forest or as Means (1997) refers to it, the ‘slope transition forest’. This ecosystem is a mix of both species from the two forests which border it (Figure 5.D). Over time the size and location of this community varies depending on the frequency and intensity of fires which reach it. In dry years with frequent fires, species of the pine savannah may dominate this region. With lack of fire, or anthropogenic fire suppression which occurred through the mid 20th century, species of the broadleaf forest encroached on these slopes and became established; however their rate of growth may be slowed on the poorer soils. In some instances following extreme fire suppression species of the hardwood forest were found on summit positions where pine savannah naturally occurs, indicating these species were invading that community.

Species within this transition zone include *P. caribaea*, gumbolimbo (*Bursera simaruba*), polewood (*Xylopia frutescens*), and several other species of hardwoods. The abundance of dumb cane (*Tripsacum latifolium*), silver palmetto (*Schippia concolor*) in particular is indicative of the savannah/broadleaf transition forest.



Figure 5. Broad ecosystems of the MPR. A: Pine forest, B: Broadleaf/Riparian Forest, C: Pine/Oak Savannah, D: Pine/Broadleaf transitional forest.

Vegetation of the Pine-Oak Savannah/Pine-Oak Forest

For the purposes of this study the vegetation of pine-oak savannah/pine-oak forest will be highlighted. Hutchison (1976) points out visitors from temperate regions find themselves in a familiar setting when traveling through the MPR. In the ecosystem's historic setting, the scattered pine trees with interspersed with oaks (*Quercus* spp.), a variety of shrubs, and a carpet of pyrogenic grasses and forbs can be compared to the pine ecosystems of the southeast United States or the pine-oak woodlands once common in the Ozarks of Missouri.

Grasses and Sedges

Grasses and sedges comprise the highest proportion of biomass of the ground layer vegetation in the historic savannah setting (Means, 1997). This physiognomic plant group is highly adapted to the presence of fire and in effect, the group facilitates fire in the MPR by providing a carpet of dry fuels with high potential of ignition during the dry season. Sites with an abundant grass and sedge component carry low intensity fires with a low to moderate rate of spread (Hutchinson, 1976). Following these fires the re-establishment of grasses has been observed to take place quickly. Hutchinson (1976) spoke of a fire which left three percent grass coverage within weeks of the fire and by one year following the fire grass had covered 24 percent of the site. Wolffsohn (1967) suggested that one year's worth of grass growth following a fire would generate enough fuel available for ignition during times of what he termed 'high' fire conditions.

The reason for the flush of grass and sedge growth following a fire is two-fold. First, the roots of grasses are protected from the low intensity fires which commonly occur in pine

savannahs. These roots utilize stored nutrients to provide for renewed growth just below charred portions of the plant. Secondly, grasses and sedges which are adapted to fire-dependent ecosystems typically produce seed at times when a good seedbed is available, such as following a fire (Means, 1997). Presumably, a good seedbed will be available following these late dry season fires which would lead into the wet season following June encouraging grass recruitment from seed.

The low-growing bunch grasses of the MPR resemble the wiregrass (*Aristida* spp.) species of the Southeastern U.S. with their rolled leaves (Means, 1997). A complete inventory of grass species of the MPR is not available; however Means (1997) identified *Axonopus purpusii*, *Mesosetum filifolium*, *Paspalum plicatum*, and *Trachypogon plumosus* as the most common grasses encountered at the MPR (Table 1).

Table 1 Common grass and sedges of the Mountain Pine Ridge.

Common Name	Scientific Name	Family
Sedge	<i>Bulbostylis paradoxa</i>	Cyperaceae
Cutgrass	<i>Scleria bracteata</i>	
N/A	<i>Anoxopus pupussii</i>	Gramineae
N/A	<i>Mesosetum filifolium</i>	
N/A	<i>Paspalum plicatum</i>	
N/A	<i>Trachypogon plumosus</i>	
Dumb Cane	<i>Tripsacum latifolium</i>	

The carpet of bunch grasses serve a vital role in terms of soil conservation by intercepting and reducing impact from rain droplets during the intense downpours of the rainy season.

One problem that can be observed on several sites across the MPR is the lowered abundance of the grass/sedge component caused by canopy closure due to fire suppression and the following initiation of dense stands of pine with a hardwood

understory. The SPB outbreak eliminated the dense pine component exposing the understory vegetation such as tiger fern (*Dicranopteris pectinata*), shrubs, and hardwoods which developed below pine stands during the era of fire suppression. The combination of inflammable dead pine trees and the understory vegetation which developed out of fire suppression carried fires that were more intense than that of the natural grass fires. The grasses, which had been suppressed for decades in these sites, were no longer locally abundant enough to support much biomass. In some sites, the lack of grass recruitment left the soil vulnerable to erosion. Several centimeters of soil loss was observed in some sites due to the loss of vegetative grass coverage (Figure 6). With the loss of surface soil horizons these sites may become vulnerable to establishment of tiger fern (*Dicranopteris pectinata*) which can carry fires more intense than that of a site dominated by grasses.



Figure 6. Limited grass recruitment following an intense fire. Grasses once occupied the majority of the ground layer vegetation in the MPR. In some locations grass recruitment following intense fires is spotty due to shading from dense stands of pine and hardwood which developed from fire suppression.

Tiger Fern

Dicranopteris pectinata (Gleicheniaceae) is a trailing or scrambling fern found in the New World tropics (Figure 7). Tiger fern, as it is more commonly known, is a pioneer species that colonizes nutrient poor or disturbed sites and can form into nearly impenetrable thickets exceeding 2m in height. In the MPR, prior to the SPB outbreak, tiger fern was restricted as ground cover beneath *Pinus oocarpa* stands in the eastern portions of the MPR and on the foot slopes within the slope transition as vegetation graded towards hardwood dominance (Means 1997, Ennion 1995). Penn et al. (2003) classify tiger fern within a 'scrubland' vegetation type due to its scattered nature and considers it a successional species with a degree of persistence being governed by topography and geology. Ennion (1995) states that this fern is prominent on disturbed sites such as logging decks, skid trails, and along roads. However, following the SPB outbreak of 2001, the occurrence of the fern became more widespread as pine mortality led to a more open condition, possibly with the initiation of tiger fern. The soil disturbance created by increased logging activity to salvage beetle killed pines may have also favored the expansion of tiger fern across the MPR. Regeneration the fern persisted beneath stands of dense pine, creating dense thickets limiting silvicultural practices such as thinning.

Tiger fern can become quite flammable during the latter half of the dry season between March and June. The fern is a very porous fuel with a high surface area to volume ratio allowing for the inflow of oxygen and ease of ignition. The fern has been observed reaching heights of 2m, which during conditions conducive to intense fires, can lead to crown scorch and pine mortality in pole size of 4-10cm (3-5 inches) stands.



Figure 7. Tiger fern. Tiger fern (*Dicranopteris pectinata*) (bright green foliage above) can form dense thickets in pine stands and can carry intense fires during the dry season.

Shrubs

The shrub layer is an important component of the MPR savannah (Table 2) (Hutchinson, 1976). This physiognomic group's diversity is dominated by individuals within the Melastomataceae family (Means, 1997). There are at least ten species in the MPR (Miller, 1997). The Melastomes, a large family of tropical plants are readily identified with leaves bearing five prominent veins or ribs. *Clidemia sericea* and *Miconia albicans* are two common Melastomes which occur throughout the MPR. The fruits of the melastomes are an important food source for wildlife including bats within the family

Phyllostomidae which contribute to the pollination and distribution of these plants (Miller & Miller, 2002). Larger specimens of the melastomes are found in the fire protected riparian hardwood forests but they will readily establish upslope, which indicates these species are encroaching upslope in regions where fire is periodic (>3 year return interval) (Means, 1997). Following the end of the dry season these plants can become flammable, especially on sites which are limiting in terms of water availability.

One of the most common species of shrub seen in the pine forest is St. John's Wort (*Hypericum terrae-firmae*). The shrub can grow up to 1.5 meters and is readily identified by its bright yellow flowers (Means, 1997). Fires will readily kill the above ground stems but the plant is often observed re-sprouting rapidly following fires. In addition to the two afore mentioned groups of shrubs, a variety of other species exist, however, the melastomes represent the most common group.

Table 2 Common Shrubs of the Mountain Pine Ridge

Common Name	Scientific Name	Family
N/A	<i>Miconia albicans</i>	Melastomataceae
N/A	<i>Miconia spp</i>	
Hairy Leaved Melastome	<i>Clidemia sericea</i>	
N/A	<i>Clidemia neglecta</i>	
N/A	<i>Tococa guianensis</i>	
St. John's Wort	<i>Hypericum terrae-firmae</i>	Guttiferae
N/A	<i>Hypericum spp</i>	
N/A	<i>Clusia salvinii</i>	

Hardwoods

For the purposes of this study, hardwoods include broadleaf trees which can achieve a dbh greater than 12.7cm (>5''). Means (1997) identifies a variety of hardwoods in his report (Table 3). The degree of historical hardwood abundance in an upland setting of the pine-oak savannah/woodland forests, where fire frequented, was most likely scattered but still maintained a significant presence overall (Means 1997). However, following the suppression of wildfire in the 1940's, species within the hardwood group readily established beneath dense stands of pine (Means, 1997). Some of these species appear to be moderately tolerant to periodic fires. Oaks in particular (*Quercus* spp.) are prominent components of the MPR and can tolerate low intensity fires and may re-sprout following intense fires. In an annual fire regime however, these species can be replaced by a community dominated by grasses and *Pinus* spp. Species of oak mentioned in Means (1997) as well as in Hutchinson (1976) include *Quercus oleoides*, *Q. peduncularis*, *Q. hondurensis*, and *Q. perseifolia*. Means (1997) is quick to point out that knowledge of how many species of oak which occur in the MPR, their ecology, and their taxonomy is meager and additional research is needed.

Another common hardwood is craboo (*Byrsonima crassifolia*). These trees can be spotted within the pine forest as well as within the hardwood stringers at the foot slopes. A variety of other hardwoods occur within the pine forests, however like the Oaks and craboo, their presence may have increased following an era of fire suppression. As with the shrub group larger specimens of these species are often found in fire protected locations such as in riparian zones. One practice which makes an assessment of the

ecological response of these species difficult is the silvicultural practice of selecting against these species through thinnings to promote pine.

Table 3 Common hardwoods of the Mountain Pine Ridge

Common Name	Scientific Name	Family
Black Oak	<i>Quercus oleoides</i>	Fagaceae
White Oak	<i>Quercus peduncularis</i>	
Red Oak	<i>Quercus perseifolia</i>	
Craboo	<i>Byrsonima crassifolia</i>	Malphigaceae
Polewood	<i>Xylopia frutescens</i>	Annonaceae
Clethra	<i>Clethra hondurensis</i>	Clethraceae
Wild anatto	<i>Bixa orellana</i>	Bixaceae

Pine

The pine forests of Central America cover 45,000km², primarily in Honduras, Nicaragua, and Guatemala, but in smaller areas within Belize (Kemp, 1972). *Pinus caribaea* var. *hondurensis*, the Central American variety of Caribbean pine, ranges from the coastal plains of Belize and portions of the MPR, through the eastern lowland of Honduras, and northeastern Nicaragua (Myers, O'Brien, & Morrison, 2006). Two species of pine occur in Belize: Caribbean pine (*Pinus caribaea*) and Honduran pine (*Pinus oocarpa*). Caribbean pine is considered a lowland pine rarely occurring above 700 meters in elevation while Honduran pine occurs at higher elevations (600-1,000 meters) (Hutchinson, 1976). This study focuses on Caribbean pine.

The taxonomy of these two pines, especially Honduran pine is often not clearly described in the literature. Complicating this issue is the hybridization of Caribbean pine and

Honduran pine, which has been observed in the interior of Honduras and in the interior mountains of Belize (Mirov, 1967). Caribbean pine is physiognomically similar to slash pine (*P. elliottii*), and in older literature such as Standley and Record (1936), they were both referred to as *P. elliottii*.

Caribbean pine is adapted to the droughty and nutrient poor soils of the MPR. Caribbean pine's thin needles, which minimize the excess loss of water through evapotranspiration, a trait common among most conifers allowing them to persist in xeric sites. Both Caribbean pine and Honduran pine have rooting systems with a high degree of horizontal growth which is concentrated within 30 cm of the soil surface allowing for the efficient capture of water through mist or dew (Dubbin, Penn, & Hodsona, 2006). These pines also exhibit a higher root to shoot ratio than many hardwoods, which persist in the nutrient rich soils derived of limestone in the surrounding broadleaf forests, relative to the acidic soils of the upland pine forest.

Pines generally are adapted to the presence of fire and in Belize they depend upon the presence of fire for regeneration (Miller A. G., 2007). Thick fire-resistant bark protects the vascular cambium from the heat of fire events. Growth form is another important trait in mitigating damage by heat as mature pines have tall wide branching canopies allowing for the dissipation of heat. The abundance of Caribbean pine on the landscape is a balance between varying fire regimes.

When fire was removed from the ecosystem, the pine stands grew dense with pine and developed a hardwood and shrubby understory which limited the recruitment of additional pine regeneration. Caribbean pine are shade intolerant requiring a disturbance event to open the canopy allowing for adequate light exposure. In the absence of fire

following the initiation of abundant pine regeneration, the pine may develop in an overstocked condition leaving the stand susceptible to drought and insect infestation. Such was the case in 2001 when highly stocked and therefore stressed pine stands succumbed to the SPB outbreak. Fires act as a thinning mechanism allowing for greater growing space which in turn allows for pine to develop under less stress within the nutrient poor, and at times droughty environment of the MPR.

Conversely with a situation in which fire is too frequent, such as bi-annual fires, pine regeneration could be nearly eliminated over time. Pine seedling and sapling mortality depends on several factors including seasonality, intensity, and the position relative to the flaming front. Caribbean pine seedlings over 1.5 meters in height can survive moderate burning conditions such as backing and flanking fires (Myers, O'Brien, & Morrison, 2006). However, high intensity fires, such as a late dry-season fire in May, can cause high mortality to most saplings less than 1.5m and may torch the crowns of taller pines.

Past Forest Management in the Mountain Pine Ridge

The greater MPR ecosystem totals 78,000 hectares (192,000 acres). The MPR Forest Reserve, an area of 43,000 hectares (107,000 acres), was established in 1944 in order to protect and to actively manage the pine forest. Private landholdings comprised the remaining lands of the MPR ecosystem. The forest had been logged prior to and following the establishment of the reserve for pine (*Pinus caribaea*). Following the stewardship of the Belize Forestry Department, a long term plan was initiated to suppress fire through construction of a network of firebreaks and access roads. The objective of fire suppression was to increase the stocking of merchantable pine. Practically the entire

reserve burned in 1949, demonstrating the lack of resources available to the forestry department at that time (Johnson and Chaffey, 1973). With the acquisition of mechanical equipment in the 1950s fire suppression became more effective. The equipment aided in the construction of roads which could be used as a barrier in the event of a wildfire.

Over the following decades, the majority of MPR communities transitioned from a savannah condition sparsely populated with pines and carpeted with rich groundcover flora of grasses and forbs to a condition in which the forest became densely stocked with pine regeneration. With time and fire suppression, hardwoods and shrubs began to populate the understory below the pine. By seeding in upslope from lower slope position where they naturally protected from fire, hardwoods became more abundant in upland settings (Means 1996). Over the next two decades, dense pine stands developed with a hardwood understory and devoid of the once diverse ground flora. Fire, which occasionally occurred in the region, no longer occurred at a frequency to produce a savanna condition. In the absence of periodic (3-5 year) fires, broadleaved species, notably oak (*Quercus* spp.) replaced the pine component in the understory. (Johnson and Chaffey 1973).

The hurricane of 1961 (Hattie) leveled large sections of the forest, in particular the more mature age classes (Friesner, 1993). Following the destruction salvage logging that ensued, and by 1973 little harvestable pine remained in the MPR according to Johnson and Chaffey's 1973 forest inventory. The next few decades leading up to the 21st century was a time of natural regeneration with minimal management, although efforts were made towards implementing prescribed fire as well as wildfire suppression during the late dry season (Headley 2005). Furthermore, the current cohort of regeneration which

initiated following the SPB outbreak vary from dense stands of pine (>10,300 trees /ha) to sparse regeneration. Current stocking conditions are an artifact of the amount of seed produced from mature seed trees prior to their mortality (Headley 2003). This leaves a portion of surviving pine in the region in the younger regeneration cohorts susceptible to fire due to the fuel build up below some stands.

Until recently, periodic fires within the pine ridge were not considered a threat to the broader MPR ecosystem. The recent SPB outbreak, the first in 50 years, changed this scenario; by virtually eliminating 90% +/- of the mature pine and placing the remaining pine regeneration at risk to fire related mortality. Following the beetle outbreak, down woody debris fuel accumulated from pine mortality. Thickets of tiger fern (*Dicranopteris pectinata*) and dumb cane (*Tripsacum latifolium*) also increased in abundance following the beetle outbreak, both of which facilitate intense fires during the dry season.

Managing Caribbean pine forests in the MPR for lumber production as well as ecological benefits may be challenging given the frequent occurrence of disturbance within the system. The 3.8 million hectares of native pine (*Pinus* species.) forest in Central America have long been threatened by a variety of destructive agents including hurricanes, droughts, wildfires, human activities and forest pest outbreaks (Billings, et al., 2004).

Southern Pine Beetle Outbreak and Response

From 1999 to 2002 the MPR was devastated by an outbreak of pine bark beetles (Coleoptera: Scolytidae) (Figure 8). This was the first major outbreak of bark beetles reported in the history of the Forestry Reserve and perhaps in all of Belize (Haack, Eckelmann, & Green, 2000). Haack et al. (2000) mention that none of staff of the Belize

Forestry Department has ever witnessed such an outbreak of SPB at such a large scale. The scale of the outbreak may be attributed to the loss of diversity and creation of a monoculture following fire suppression and the recruitment of dense stands of pine. The bark beetle infestation led to mortality to approximately seventy percent of the MPR and an estimated, but unconfirmed ten to fifteen percent of pine forests of the South Coastal Plains of Belize (FAO, 2008).

Billings et al. (2004) attribute the SPB outbreak to an abundance of dense, susceptible pine stands, a failure to recognize and respond to the beetle outbreak in the early stages, and a severe reduction in forestry personnel in 1995. Haack et al. (2000) attributes a severe drought in 2000 just prior to the beetle outbreak as an inciting factor, as drought has triggered outbreaks of bark beetles across a variety of ecosystems (Mattson & Haack, 1987).

Twelve species of pine bark beetles native to Mexico and portions of Central America have been identified (Billings et al 2004). Haack (2000) identified three common pine bark beetle that are native to Belize: *Dendroctonus frontalis* Zimmermann, *Ips calligraphus* (Germar), and *Ips grandicollis* (Eichhoff). FAO (2008) identifies the same beetles mentioned by Haack (2000) in addition to *Ips apache*. During the outbreak Haack (2000) set six multiple-funnel traps baited with frontalin and trapped 1,934 *D. frontalis* adults, indicating they were the primary agent of the outbreak. In addition to *D. frontalis*, 153 *I. grandicollis* adults and six *I. calligraphus* adults were found in the traps. Billings et al (2004) mentions *D. frontalis* and *D. adjunctus* (the latter primarily in Guatemala) are the most destructive bark beetles in the region. Midtgaard and Thunes

(2002) indentified a new species of *Dendroctonus* in Central America, but the role of this new species in the recent outbreak remains unclear. The taxonomy of these beetles may need additional observation as ecologists in the region have suggested these may be subspecies of bark beetles encountered in the Southeastern United States.

While it appeared that 90%+ of the pine timber that existed before the beetle epidemic has died, there were pockets of living pine throughout the MPR in 2005 (Mark Meagher personal communication). During visual observations on the eastern boundaries of the MPR, some trees were observed as infested, however it should be noted these were individual trees and not pockets of trees infested. It is assumed that small populations of the SPB exist throughout the ecosystem and will again reach epidemic densities given recurrence of drought conditions and the absence of forest management. Following the outbreak, many managers and landowners in the region realized the potential for loss of the pine component due to the unprecedented bark beetle outbreak. They also feared that intense wildfires would occur following the outbreak which would eliminate any developing pine regeneration. However, observations of pine regeneration abundance have exceeded expectations. In some sites where fires occurred prior to the SPB outbreak, dense stands of pine regeneration have initiated, which require immediate thinning to promote tree vigor and prevent an additional beetle outbreak.



Figure 8. The southern pine beetle. The southern pine beetle (*Dendroctonus frontalis*) highlighted by the red circle above, was found on a mature pine on the Eastern portion of the Mountain Pine Ridge. Southern pine beetles exist in low density across Central America but reached epidemic proportions in 2001 and attacked overstocked stressed pines causing over 90 percent mortality across the MPR.

Fire Ecology and Management in the Mountain Pine Ridge

It is likely that in the Mountain Pine Ridge and the other upland pine savannas, the balance between savanna and broadleaved forest is determined primarily by fire, caused naturally by lightning and also by human sources. Fires can occur any time during the year as long as fuels have experienced desiccation to the point where ignition may occur. The greatest incidence of fires is within the dry season, from January to May, a period of limited precipitation and low humidity. Fire risk becomes particularly extreme in May following several months of limited precipitation and the incidence of storms which are accompanied with lightning and no rainfall.

Forestry Department data shows that from 1963-1970 63% of reported fires in the MPR were caused by lightning with the remaining 37% of fires being caused by humans (Meerman & Sabido, 2001). With the increase in population within the region in the past 40 years it can be expected that anthropogenic fires have increased, however lightning caused fires are still common.

The degree to which anthropogenic fires occurred in the MPR during Mayan occupation (7,000 B.P.-1500 CE) has been subject to debate. Johnson and Chaffey (1973) assert that the proximity of several Maya centers off the MPR escarpment on richer soils dominated by limestone may have had some effect on the ecosystem. With its resinous wood, Caribbean Pine makes an excellent fuel wood and was used in a variety of ceremonial practices (Wyatt, 2008). Wyatt (2008) found evidence pine refuse from household hearths was used as a soil amendment suggesting extensive use of pine by the Maya village, one of which existed in Chan, an area where pine is a non-local species amongst calciphile hardwood species. The evidence of pine remains within sites where pine did not occur, coupled with the proximity of Maya centers surrounding the MPR suggest anthropogenic fires may have frequently occurred in the region. With the highest population and most intensive land use occurring around 670 CE, and with evidence of human habitation occurring at least 10,000 years B.P., it can be possible that humans could have influenced fire frequency on the MPR landscape for millennia.

Regardless of the degree of anthropogenic or natural influence in the ignition of fires, observers in the early 20th century found an ecosystem shaped by and dependent on the disturbance by fire. There is little information regarding the condition of the MPR prior to the late 19th and early 20th century. One of the earliest written accounts was by Hooper

(1887) in which he visited a portion of what he termed, 'the Great Southern Pine Ridge'. Standley and Record (1936) describe the MPR with a rich herbaceous layer and sparse pine which can be attributed to periodic fire. Anecdotal accounts by local hunters report pursuing game on an open landscape with sparse pines, a condition far different than the present landscape dominated by scattered dense patches of tiger fern dense stands of *Pinus* spp. regeneration, with some savannah-like grasslands.

Initial efforts of fire suppression were focused within the granite basin of the MPR. The periphery of the granite basin, such as the highly dissected region east towards the bald hills was relatively inaccessible, leading to containment rather than suppression in these areas (Hutchinson, 1976). The primary goal of fire suppression by the Forestry Department in the MPR was to increase the stocking of pine timber across the pine savanna (Means, 1997). The practice worked as planned; the savannah condition was replaced by dense stands of pine and the groundcover flora of grasses, sedges and forbs were shaded out as canopy closer increased. In effect, fire suppression in the MPR was too 'successful' in producing vigorous dense stands of pine (Chun, 2005). Hardwoods established beneath dense stands of pine. These hardwoods and remained down slope in natural fire protected sites and scattered in low numbers in the pine savannah.

Historically, the primary fuels of the MPR were grasses and sedges and to a lesser extent shrubs as well as pine and hardwood leaf litter (Means 1997). Ignition potential of grasses and sedges increases following periods of limited precipitation. Grass fires are generally 'cool' fires in that they have a moderate rate of spread limiting the time the fire is present on the surface. Hunt (1962) describes the rate of spread from a lightning fire at 9 meters per minute. Wolffsohn (1967) calculated a rate of spread with a fuel bed of

grasses at 1.6 km/hour (1mi/hour) and estimated from observations that fires within dry scrub and fern moved at twice the speed of grass fires. Following the fires the grasses reestablish quickly.

Presently, the fuel bed composition has been altered from its historic composition of primarily grasses and scattered shrubs. In dense stands of pine regeneration hardwoods, shrubs, and ferns are becoming established as they had during the 1950's, 60's and 70's following fire suppression. Fires composed of fuels such as shrubs, hardwoods, and ferns are deemed 'hotter' with a higher rate of spread and likely higher fire intensity than that flora groundcover of grasses and sedges (Figure 9).

Also increasing in abundance following the SPB outbreak is tiger fern (*Dicranopteris pectinata*). In other areas, pine is absent as the seed trees were eliminated by the SPB and fires following the outbreak eliminated what little pine regeneration managed to become established.



Figure 9. Wildfire in the MPR. Wildfires are common in the MPR, especially during the end of the dry season towards April/May. Photo: George Headley.

Goals of Research

Few ecological studies have taken place in the pine-oak forests of the MPR. The most pressing management issue in the MPR is implementation of an appropriate fire management regime to facilitate quality timber production, while at the same time, emulating natural fire regimes that support the biodiversity of the ecosystem. In order to advance the productivity of the ecosystem, managers must implement a long-term fire management plan that takes into account the response of the ecosystem following disturbance events. Research in the MPR was initiated in order to augment the ecological understanding of the ecosystem, which in turn could be used in management.

Following a 2002 workshop held at Rio Bravo Conservation and Management area, ecologists from The Nature Conservancy developed a basic conceptual model (Figure 10) illustrating the response of Belize pine savannah ecosystems to varying fire regimes. The Rio Bravo pine-savannah ecosystem is ecologically similar to the MPR, however the terrain is less dissected than that of the MPR.

The diagram indicates the effect of modifications in the assumed historic fire regime (2-7 year fires) on successional dynamics in pine savannahs of Belize. Following fire suppression they assert that the forests will develop from a savannah condition to a woodland pine-oak forest to eventually a forest where hardwoods dominate. With annual fires the pine savannah would eventually develop into a shrub/grassland.

Data from this study will aid in validating the model developed by The Nature Conservancy and will be used in the development of a new fire regime model specific to the present pine-oak ecosystems of the MPR. In the absence of fire following the SPB

outbreak, hardwoods and shrubs would be expected to increase in abundance while sites that experienced intense fire would exhibit soil loss and establishment of tiger fern.

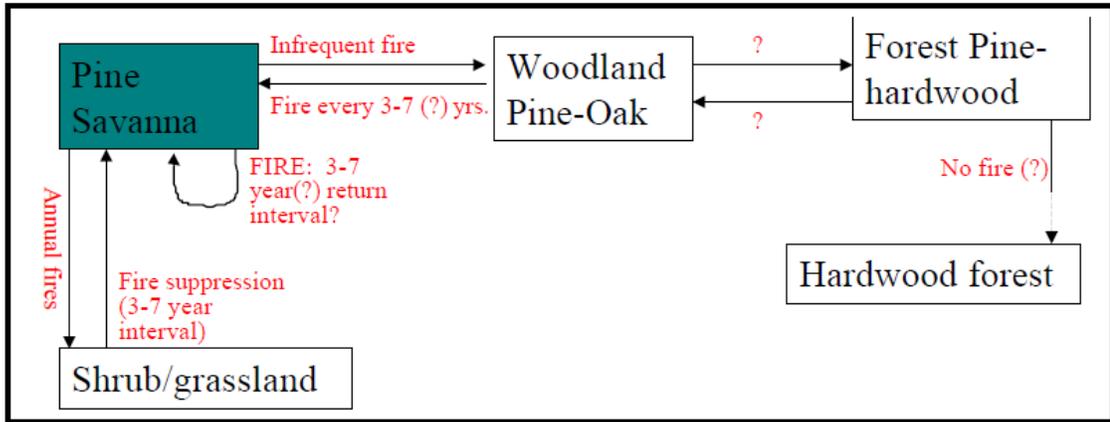


Figure 10. Fire regime model developed by The Nature Conservancy (2002) for describing successional dynamics in Belize pine savannahs

The objectives of the study were to:

1. Determine the effect of time since last fire and the short term fire frequency, on potential vegetative fuels, pine regeneration, down woody debris, litter, and bare mineral soil.
2. Determine the association of foliar canopy coverage to soil, litter, down woody debris and vegetative fuels.
3. Use GIS to determine if slope aspect plays a role in the distribution of tiger fern (*Dicranopteris pectinata*) colonies, an important fuel in the MPR.

Based on the objectives, the following null hypotheses (H_0) were developed:

H_0 1: Time since fire and short term fire frequency will have no effect on vegetative fuels, pine regeneration, down woody debris, liter and bare mineral soil

H₀ 2: Foliar canopy coverage has no effect on soil, litter, down woody debris and vegetative fuels

H₀ 3: Aspect does not play a role in the distribution of tiger fern

Study Area

The research took place on the property owned by Bull Run Overseas Ltd and as well as lands owned by Hidden Valley Inn. The properties are located in the Cayo Political District in west-central Belize. The Bull Run Overseas Ltd. (Bull Run) property lies on the northern end of the MPR. The property is bounded on the south by the MPR Forest Reserve, on the west jointly by the Pine Ridge Enclave and the Elijio Panti National Park and on the north by the hardwood dominated escarpment leading down to Barton Creek, and the Thousand Foot Falls National Park (Headley 2005). The total acreage owned by Bull Run Overseas Ltd as well as Hidden Valley Inn totals 5,862 ha (14,486 acres) (Figure 11).



Figure 11. Location of the Mountain Pine Ridge. The location and the MPR (red) and the study site within it (green).

Methodology

Sampling Site Selection

Since the late 1990's, fire events including prescribed and wildfire events were recorded and mapped by Bull Run Overseas, Ltd. These represent all fires occurring on and adjacent to the property managed by the company from 1998 until January 2010. For the purposes of research, fire units were categorized into three classifications relative to time since fire following the SPB outbreak, including recent (1-3 years since fire), midrange (4-10 years since fire) and extended (11+ years since fire) (Figure 12).

Plot selection was based on several criteria. To eliminate the effect of slope and aspect as well as variation within soil types, plots were established on summit positions on the landscape with a slope percentage of less than fifteen percent. Plots were also located in areas free of any mechanical disturbance following the SPB outbreak such as logging, thinning, skidder trails, or within 20 meters of roads. These criteria limited sampling areas to specific sites where sampling could occur.

A 100x100 meter grid was overlain on a map of the study area. The grids were used to install plots systematically within varying burn units. Within each site, plot arrays of three plots were established in a row North to South or West to East within 100 meters of each other when possible. In some locations where a plot within a row fell into a zone which did not meet criteria (e.g. conflict with a road or exceeding slope percentages of 15 percent), it was shifted 100 meters in any direction until sampling criteria were met. The plots were located with the use of a GPS device. Accuracy utilizing GPS equipment

ranged from 2 to 10 meters. Plots were sampled from January until the onset of the dry season in March when plants began to desiccate in which data would be affected.

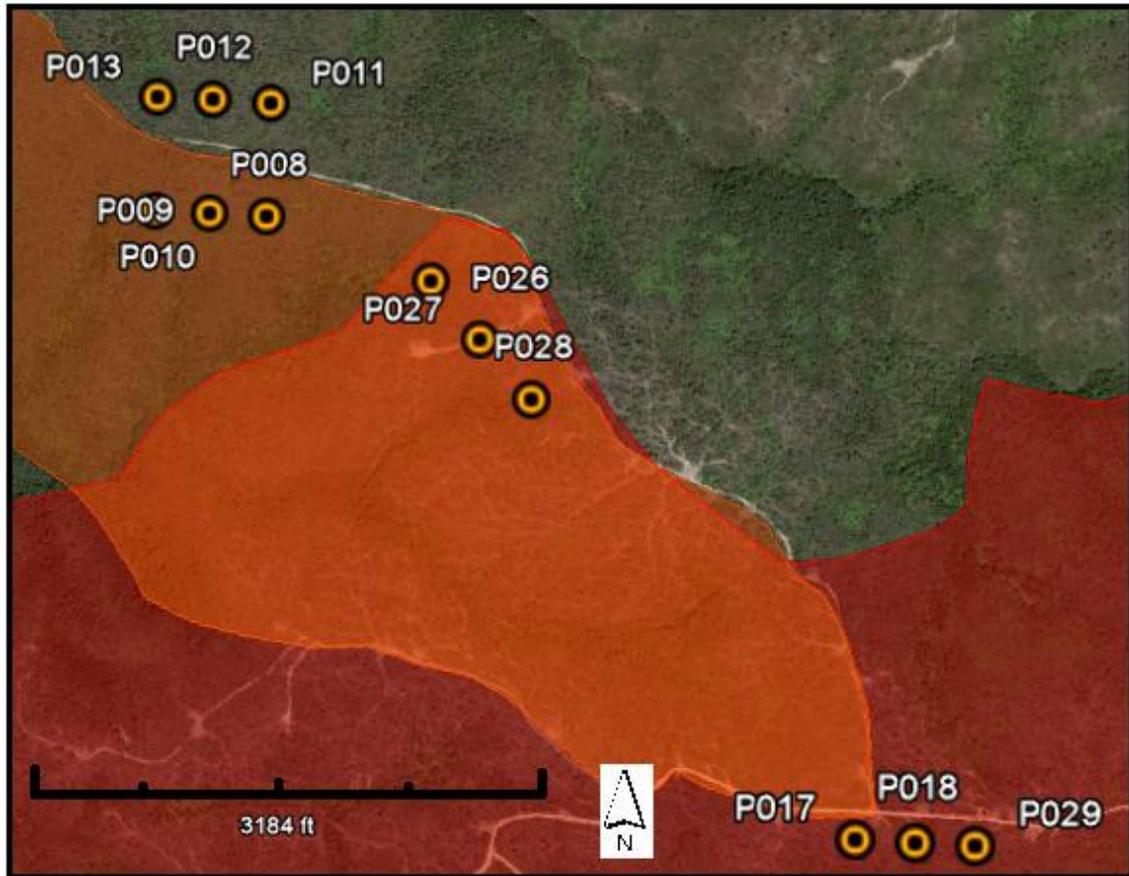


Figure 12. Example of systematic location of plots. Dark red layers represent recent fire intervals, orange represent periodic, and plots with no colored polygon represent extended fire intervals. The dark orange polygon (center) shows overlap between a recent and periodic fire event.

Plot Design and Use

Circular 1/5th of an acre fixed radius (16m/52.7') plots were used for sampling (Figure 13). Four, 16m transects were established in each cardinal direction from plot center. Along each transect four 1m² quadrats were established at 4, 8, 12, and 15 meters from plot center. Quadrats represented sampling units for estimating abundance and mean

height of physiognomic plant groups, abundance of woody debris, bare mineral soil, and litter. Percent foliar canopy coverage estimates were made at 8m and 15m along each transect for a mean plot estimate of canopy coverage based on 8 measurements.

At the terminus of each transects a 1/100th of an acre subplot and a 1/1000th of an acre microplot was established for measurement of pine saplings and seedlings respectively.

SAS was used for all statistical analyses. ANOVA, using PROC GLM was used for assessing the effect of time since last fire (treatment variables) on physiognomic vegetative group abundance, height of vegetation, abundance of litter, bare mineral soil and woody debris. SAS PROC REG was used for detecting the relationship between canopy coverage, as the predictor variable and abundance of particular vegetation physiognomic groups.

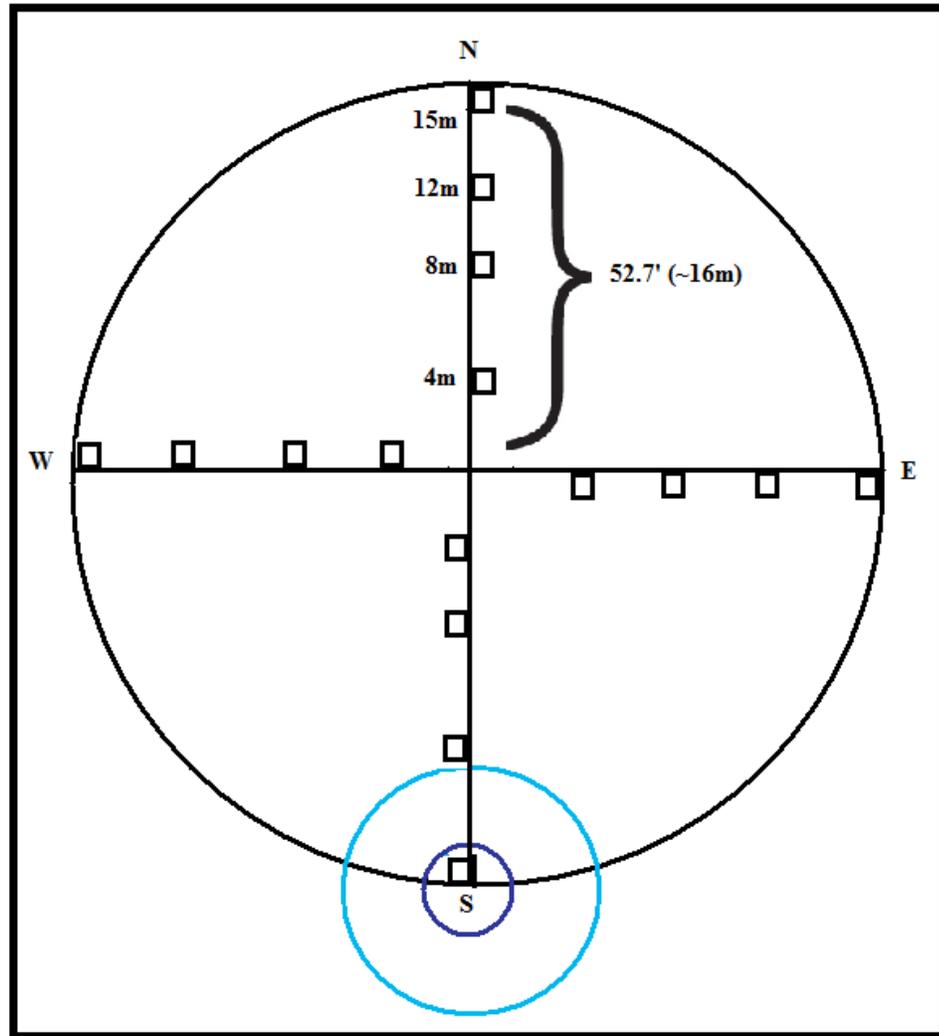


Figure 13. Plot design. Plot utilized for data collection. At the terminus of each transect a 1/100th acre plot (light blue circle) and a 1/1000th acre microplot (dark blue circle) were installed for pine regeneration surveys. Along each transect a 1m² quadrat was placed at the 4m, 8m, 12m, and 15m from plot center for assessment of physiognomic vegetative group abundance and heights.

Physiognomic Vegetative Groups

Within each of the four transects, four 1m² quadrats were established. These quadrats were established for measurements of physiognomic plant groups including grasses, forbs, ferns, vines, shrubs, hardwoods and pine. Within each of the 16 quadrats in each plot, the percent abundance of each physiognomic group was visually estimated and recorded. The mean abundance for each physiognomic group in all 16 quadrats was calculated. Height of the tallest individual within each quadrat for each physiognomic plant group was measured using a height pole. The mean height for each physiognomic plant group was calculated for the whole plot using data from all quadrates.

Down Woody Debris, Litter, and Bare Mineral Soil

Within each quadrat the abundance of down woody debris was visually estimated. Down woody debris was classified into three classes including fine woody debris (< 3'' in diameter), coarse woody debris (>3'' in diameter), and rotten woody debris which was classified as material no longer supporting its cylindrical form with the application of pressure to the material. In addition to down woody debris the percent abundance of litter and bare mineral soil (BMS) was visually estimated within each plot.

Pine Regeneration

Subplots (1/100th acre) were utilized for measurements of saplings (trees 1-5'' in diameter) while microplots (1/1000th acre) were utilized for counts of pine seedling regeneration (all pines less than 1'' in diameter) at breast height 1.37m (4.5'). Within

each plot four subplots and microplots were established. The mean basal area for each plot and the mean number of seedlings per acre for each microplot were calculated.

Canopy Coverage

Percent foliar canopy coverage of pine and hardwood trees above 2.5 meters were estimated using a convex densiometer within each plot. Canopy coverage was measured at the second and fourth quadrat site for each transect. The mean percent foliar canopy coverage estimates were later calculated. These measurements were utilized to determine the effect of canopy coverage on the abundance and height of grass, fern, forb, and vine physiognomic vegetation classes.

Analysis of Slope Aspect and Tiger Fern Abundance

Tiger fern (*Dicranopteris pectinata*) is a vegetative fuel which can initiate intense fires during the dry season. It is important to understand where and why the fern colonizes in order to manage this fuel. Tiger fern colonies are easily identified on satellite imagery as they appear as bright green patches relative to surrounding vegetation (Figure 14a). On infrared imagery tiger fern colonies appear as a bright white color relative to surrounding vegetation allowing for quick identification (Figure 14b). Using Arcmaptm version 9.3.1, GIS software and imagery provided by Geoeye Inc., the abundance of tiger fern were measured relative to slope aspect.



Figure 14. Tiger fern as observed from satellite images. Tiger fern is easily observed forming circular colonies in color images (left) and appears bright white in infrared satellite images (right).

Aspects were grouped in North (slopes with an aspect of 330-30°), East (60-120°), South (150-210°) and West (240-300°). GIS layers indicating slope aspect were not available so aspects had to be manually outlined. Nineteen slopes of each cardinal direction were identified and outlined for a total of 76 identified slopes. In each slope a 2.5 hectare square plot was situated mid-slope in the geographic center of each slope polygon as this size plot fit within all selected aspects. To avoid bias in selection, each outlined slope GIS layer had a solid fill during the placement of the 2.5ha plot layer. Once all 2.5 ha plots were placed in the geographic center of each outlined slope, the solid fill was

removed and all visible tiger fern was mapped in the plot (Figure 15). The area of tiger fern was ascertained within each plot through the use of Arcmap and divided by the 2.5 hectare plot for a calculation of the relative percentage. Tiger fern abundance data was recorded into excel and inputted into SAS where the PROC GLM procedure was used to calculate differences between slope aspects and percent abundance of tiger fern.

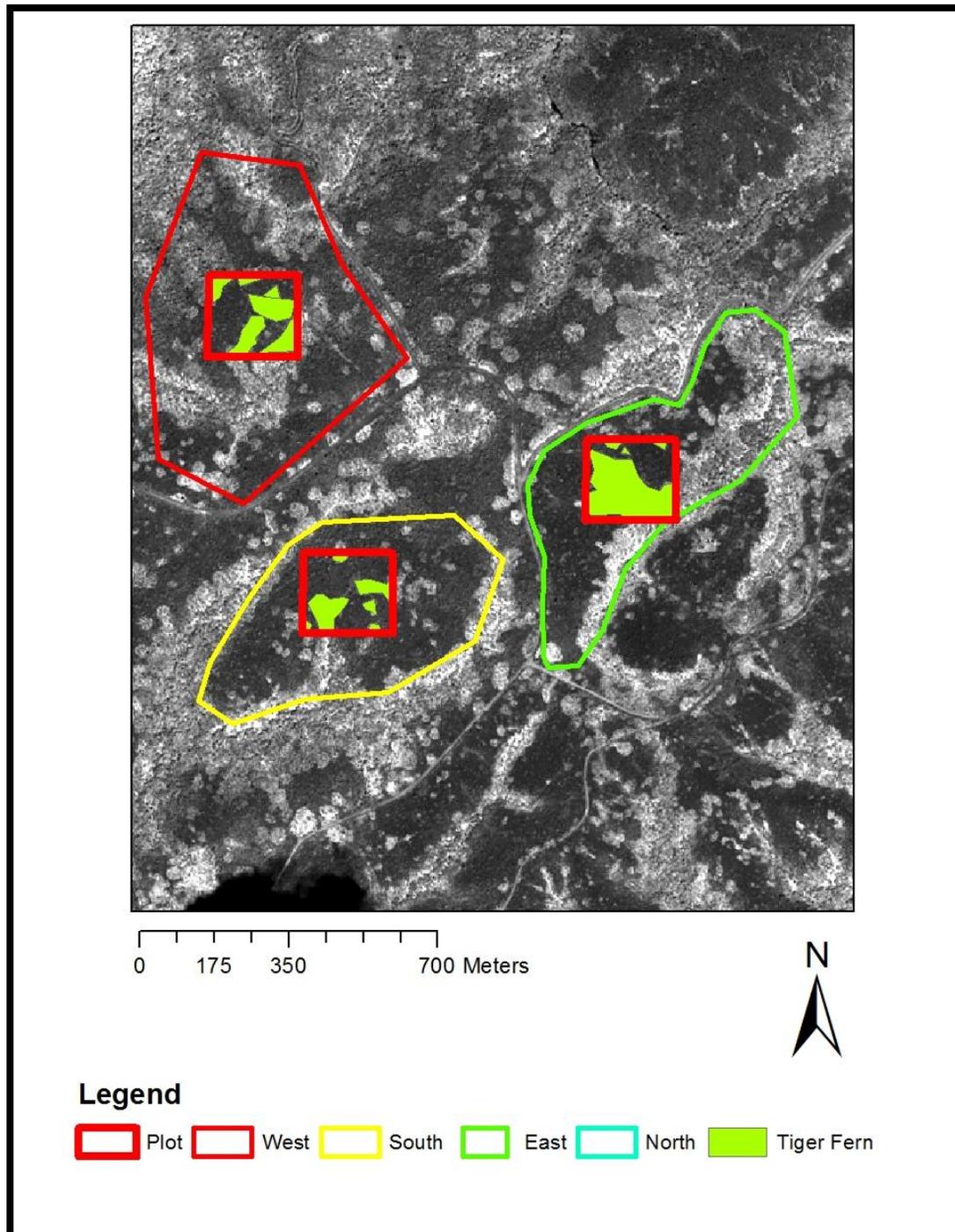


Figure 15. Sampling design for GIS analysis of tiger fern relative to slope aspect. Two and a half hectare plots (red squares) are placed mid-slope and center of each slope aspect. The percent of tiger fern occurring within each plot was ascertained using Arcmap GIS software.

Results

Time Since Fire and Vegetative Response

From late January until early March 2010 thirty nine plots were measured to assess the effect of fire temporal intervals on the abundance of grasses, fern, forbs, vines, shrubs, hardwoods and pine. Of all the physiognomic plant groups, shrubs, hardwoods, and ferns were the only three to show a significant response relative to fire temporal intervals (Figure 16). For hardwood abundance only the 1-3 and 11+ fire temporal intervals varied significantly ($P= 0.0265$) (Figure 17). In terms of hardwood heights, the 'recent' fire interval varied significantly from the midrange (4-10 year) and extended (11+ year) fire intervals ($P = .0147$) (Figure 17).

Shrub abundance was significantly greater in the extended fire events than the recent fire events ($P=0.0351$) (Figure 19). The shrub physiognomic group mirrored the results of the hardwoods with shrub height showing significant difference between the recent (1-3 year) and the midrange (4-10 year) as well as between the recent (1-3 year) and extended (11+ year) fire intervals ($P=.0002$)(Figure 20).

Fern abundance was significantly greater in the midrange (4-10 year) fire events than the 1-3 and 11+ year fire intervals ($P=.013$) (Figure 21).

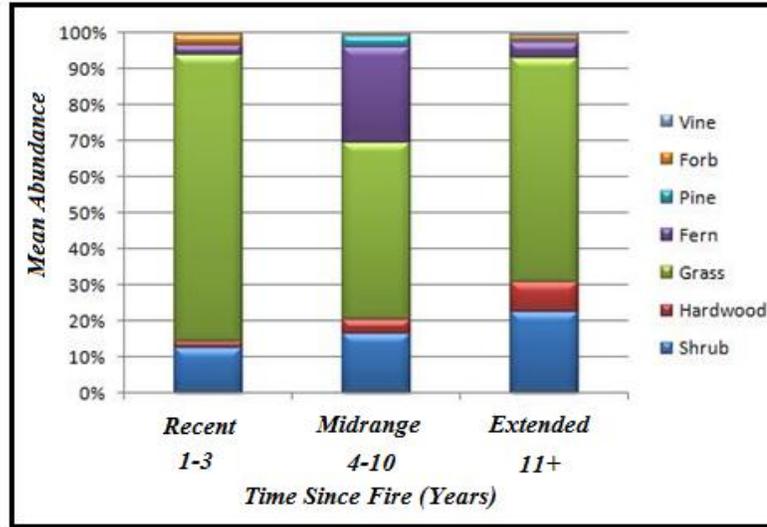


Figure 16. The mean abundance measured as percent cover of vegetative physiognomic groups in each of three temporal stages of fire activity on the MPR. The categories represent time since last fire in and recent refers to an area that has burned 1-3 years before sampling, periodic (4-10), and extended fire represents 11 or more years since the last fire.

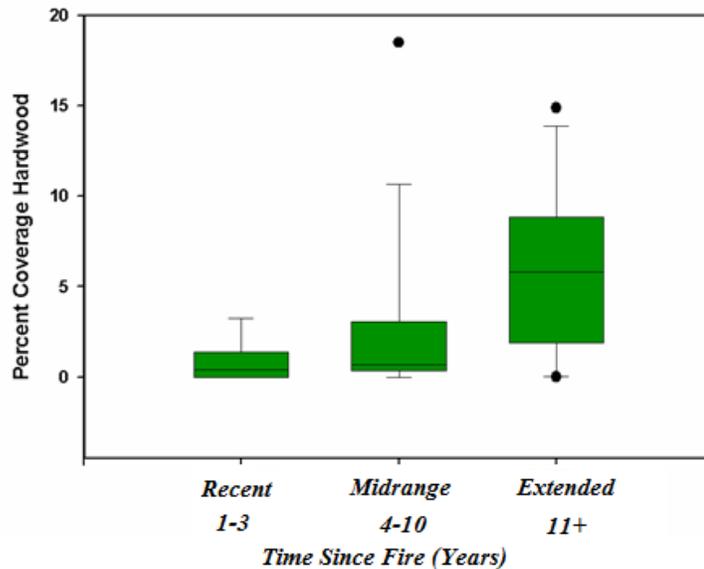


Figure 17. The percent abundance of hardwoods in each of three fire time periods fire at the MPR. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

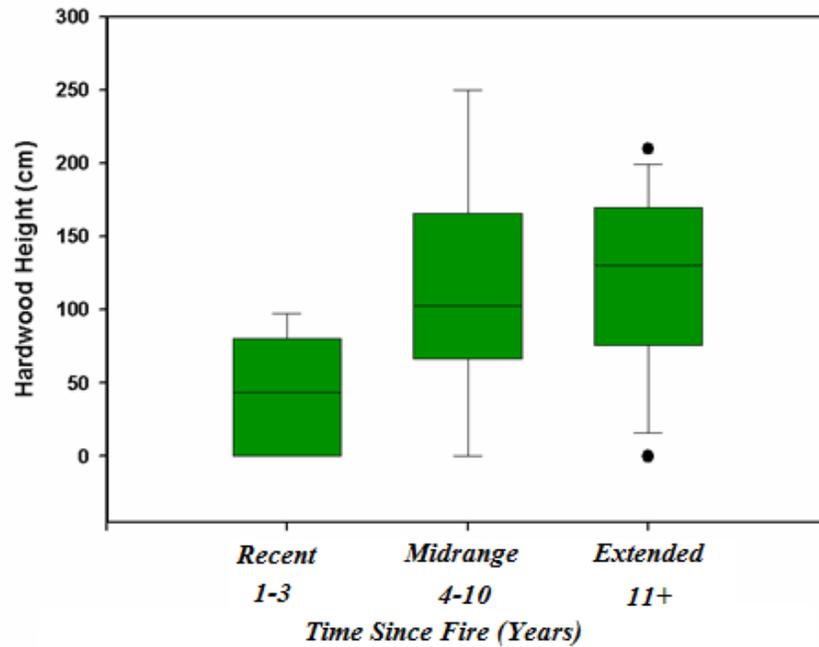


Figure 18. Hardwood average height in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

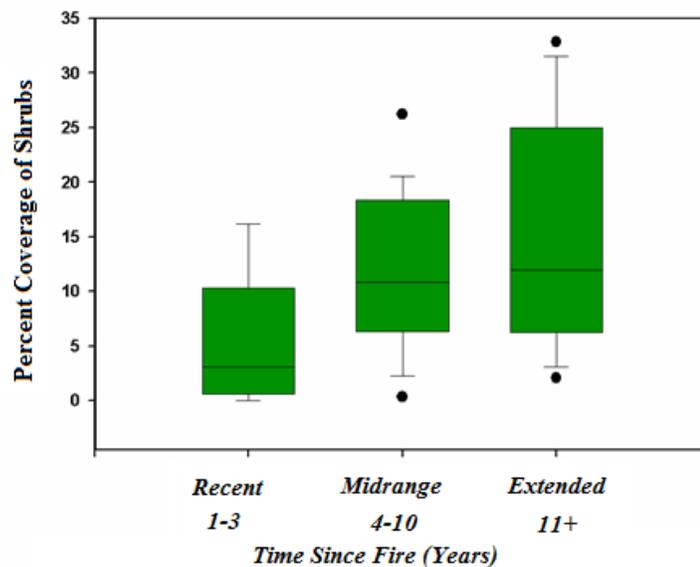


Figure 19. The percent abundance of shrubs in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

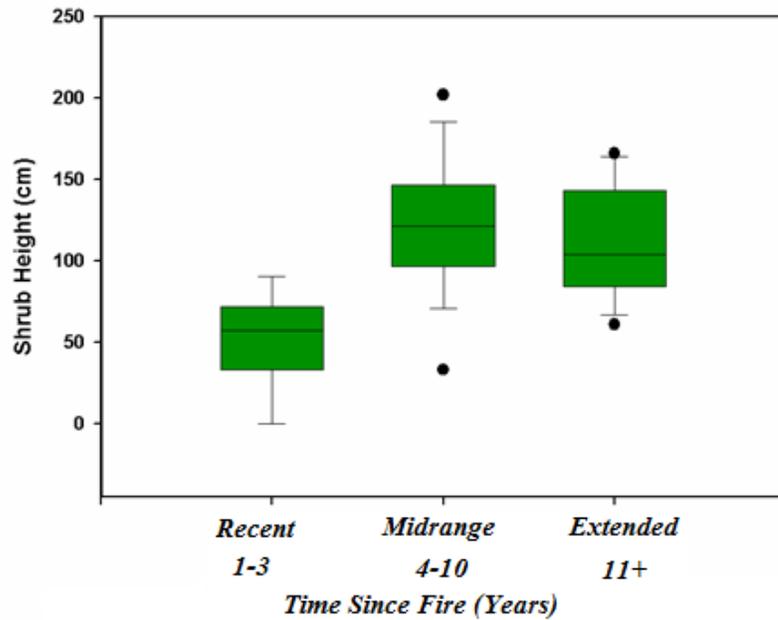


Figure 20. Shrub average height in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

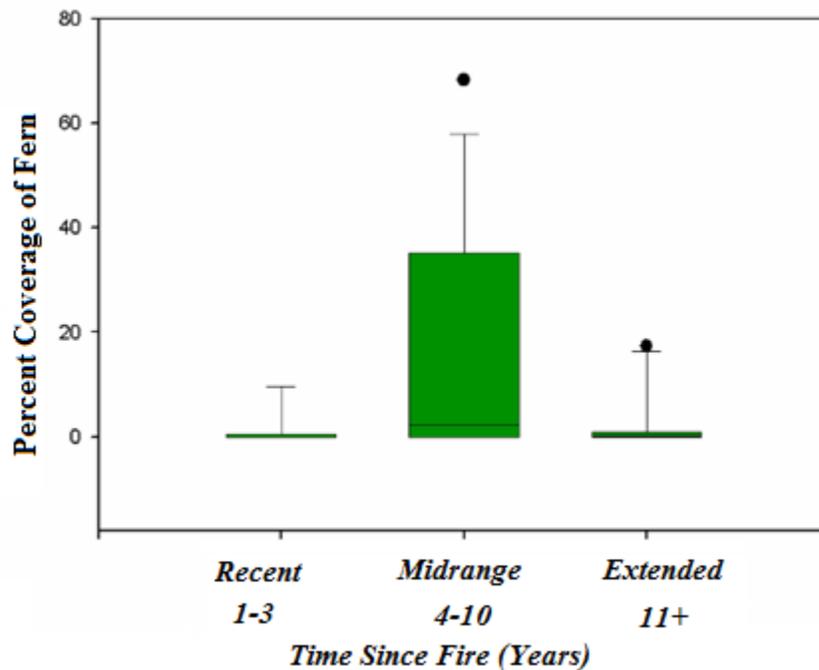


Figure 21. The percent of fern abundance in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

Time Since Fire and Pine Regeneration Response

Pine seedling density was significantly greater at the sites that burned 4-10 years ago than the sites with recent (1-3 year) events. ($P=.029$) (Figure 22). Similarly, the basal area of pole size pines was significantly greater at sites that burned 4-10 years ago than the sites with recent fire events. ($P=.0063$)(Figure 23).

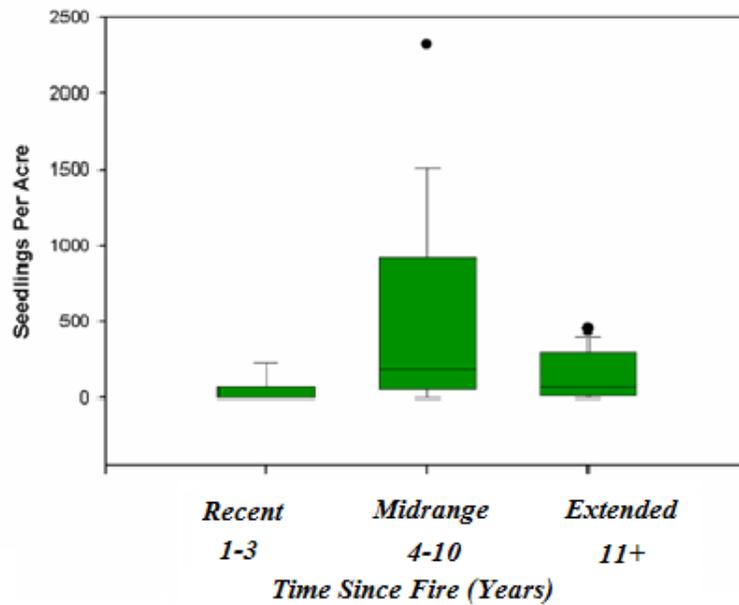


Figure 22. Pine seedlings per acre in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

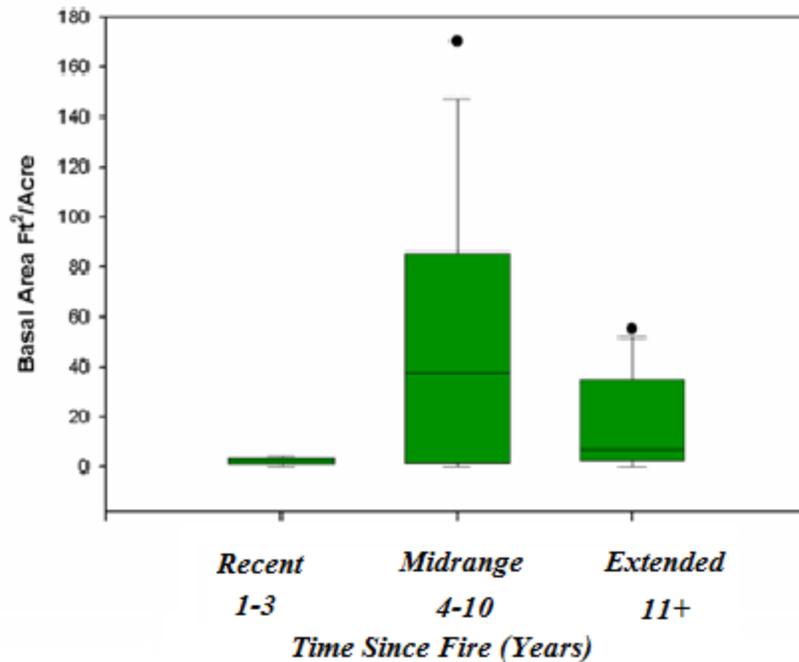


Figure 23. The basal area of pole size pines in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

Percent Canopy Coverage and Vegetative Response

Canopy coverage estimates were determined within 39 plots. The canopy coverage was compared to the abundance of all vegetative physiognomic groups. Only two vegetative groups, grasses and ferns, showed a trend relative to percent foliar canopy cover. Canopy coverage correlated with fern abundance ($R^2 = .73$) (Figure 24), but negatively and more weakly related with grass abundance. Grass abundance shows a weak negative correlation to canopy coverage and shows a noticeable decline in abundance with > 60 percent canopy coverage, ($R^2 = .20$) (Figure 25).

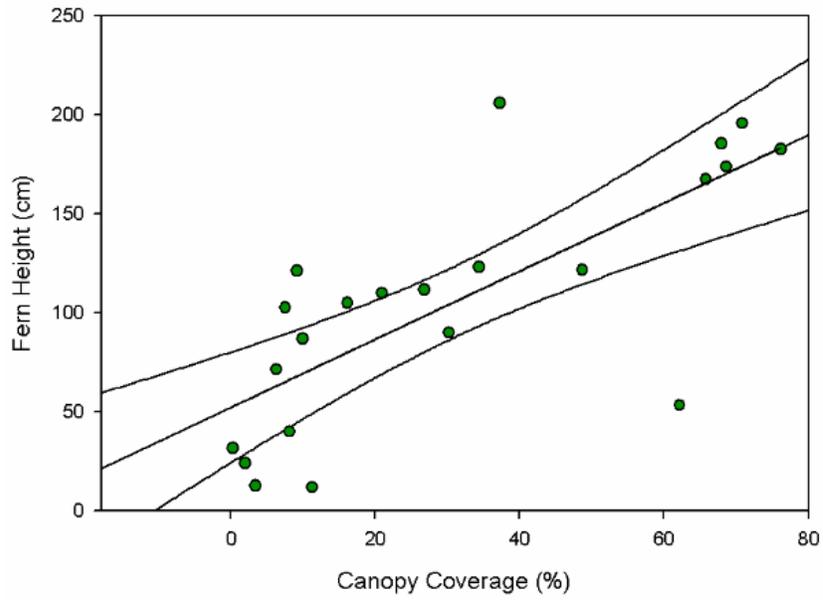


Figure 24. Regression indicating the relationship between fern height and percent canopy coverage with a 95% confidence interval. $R^2=0.73$

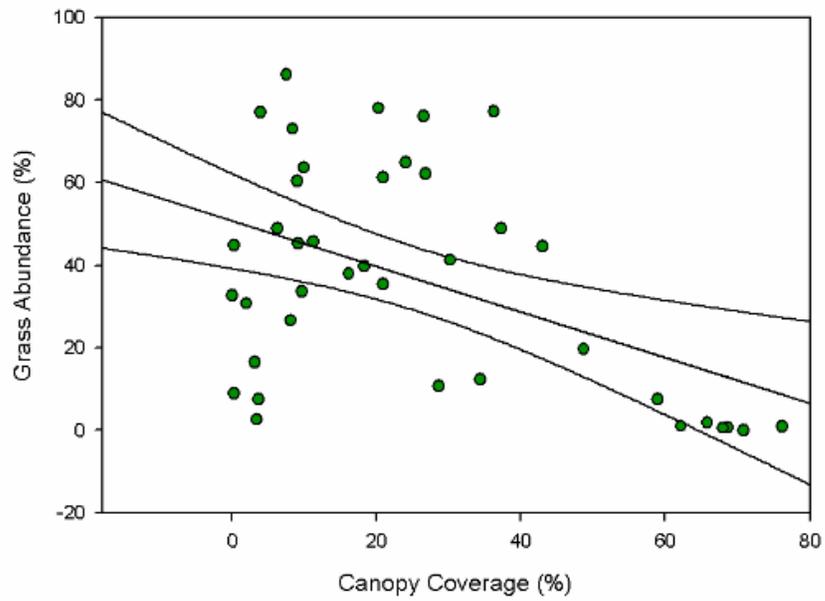


Figure 25. Regression indicating the relationship between abundance of grass and canopy coverage with a 95% confidence interval. $R^2=0.20$

Time Since Fire Compared to Down Woody Debris, Soil and Litter

No significant difference was found between any of the fire event times in relation to any of the down woody debris conditions including fine woody debris, coarse woody debris, or rotten woody debris. The abundance of bare mineral soil (BMS) as well as litter showed significant variation between the fire events (Figure 26). Specifically, there was a significantly greater amount of BMS in the recent (1-3 year) fire events than the midrange (4-10 year) fire events ($P < .0001$) as well as between the recent and extended (11+ year) fire events ($P < .0001$) (Figure 27).

The abundance of litter between plots and certain fire events also showed significant variation. A significant difference existed between the recent (1-3 year) fire interval and the midrange (4-10 year) fire event ($P = 0.005$) as well as between the recent and extended (11+ year) fire events ($P = 0.0013$) (Figure 28).

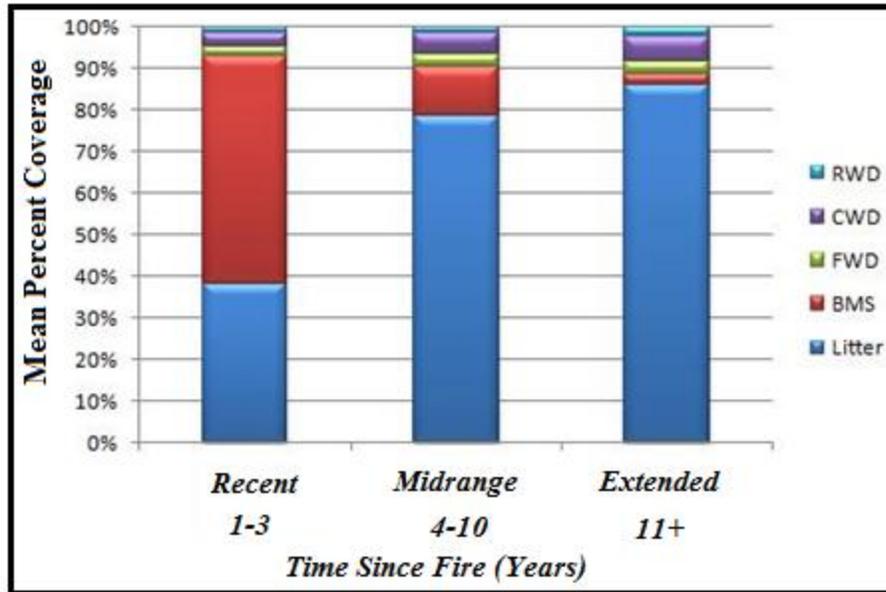


Figure 26. The mean abundance for rotten woody debris, coarse woody debris, fine woody debris, bare mineral soil, and litter relative to recent, midrange and extended fire intervals.

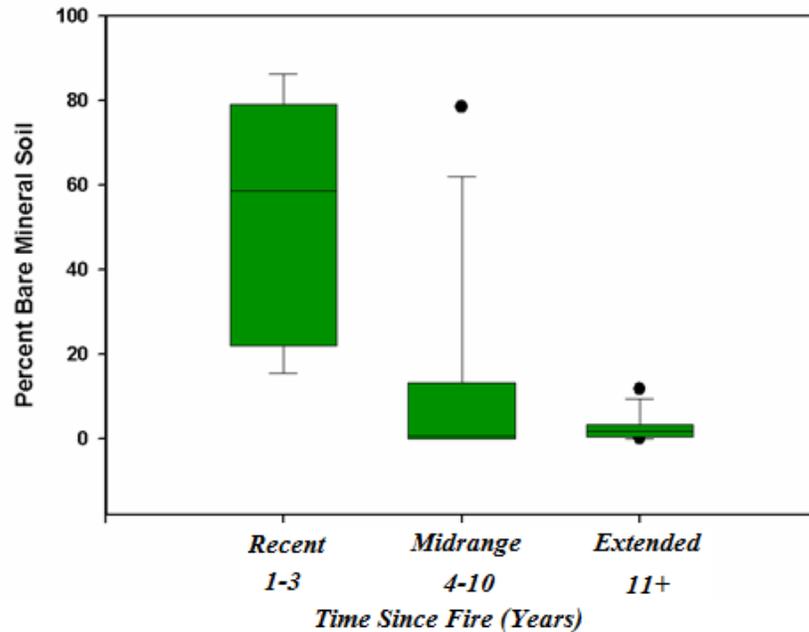


Figure 27. The percent of plot area in bare mineral soil in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

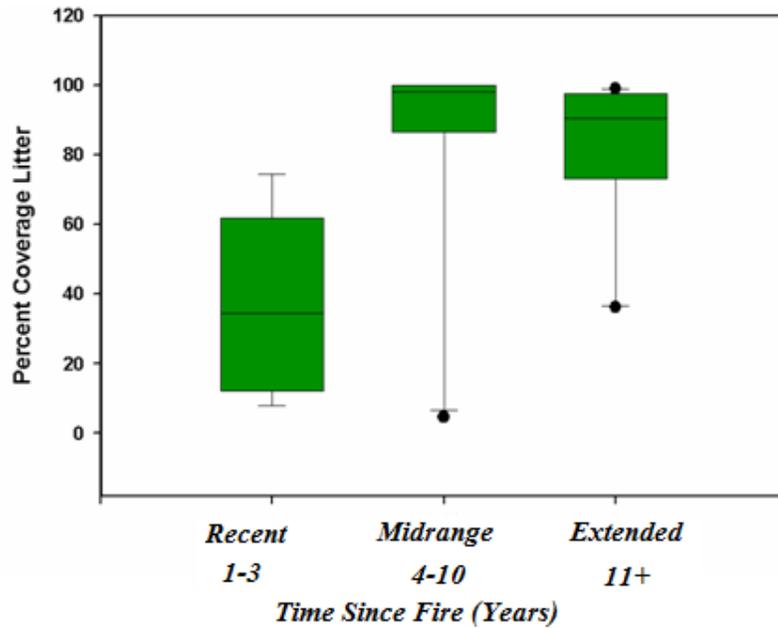


Figure 28. The percent of plot area in litter in each of three fire time periods. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

Fire Frequency Following the SPB Outbreak

No significant difference was observed between any measured variables and the frequency of fire following the SPB outbreak,. Specifically coarse woody debris (Figure 29) and the abundance of hardwoods (Figure 30) show a visual trend, however statistically, it is not significant.

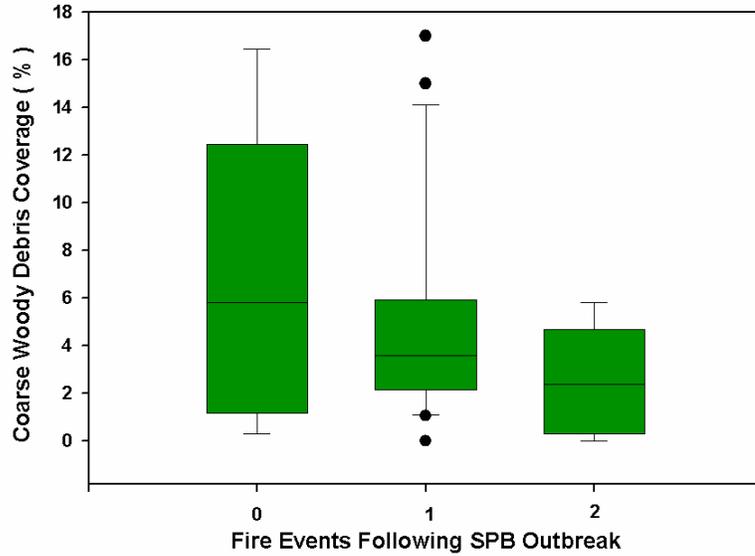


Figure 29. Percent Abundance of coarse woody debris relative to the number of fire events following the SPB outbreak of 2001. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

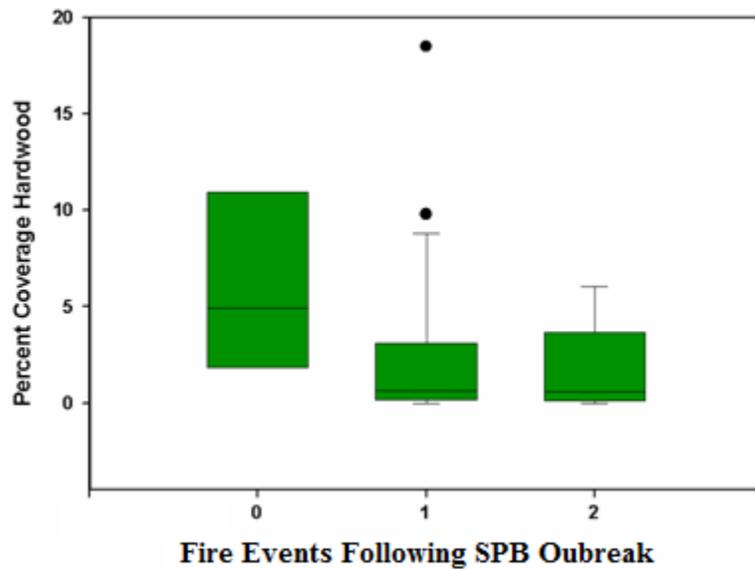


Figure 30. Percent coverage of hardwoods relative to the number of fire events following SPB outbreak of 2001. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

GIS Analysis of Tiger Fern

Data were collected from 19 plots from each aspect for a total of 76 2.5 hectare plots. The abundance of tiger fern in each plot was mapped using Arcmap. No statistically significant relationship exists between aspects and the abundance of tiger fern (*Dicranopteris pectinata*). The southern aspect showed the greatest mean abundance compared to the other aspects, however the difference is not statistically significant (Figure 31).

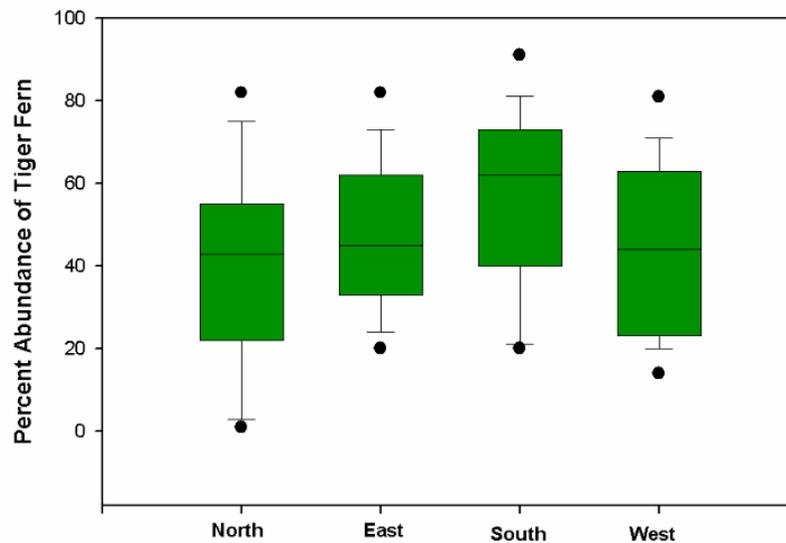


Figure 31. Abundance of tiger fern relative to aspect. The line in the middle of the box represents the median for the data, with middle quartiles filling the remaining portion of the box. The lines extending beyond the box represent the upper and lower quartiles while dots extending beyond the lines represent outliers.

Discussion

Physiognomic Vegetative Groups

Among the physiognomic vegetative groups, the hardwoods and shrubs showed the greatest response to varying times since fire. The data show that both hardwoods and shrub abundance declines in the more recent fire intervals relative to extended fire intervals. The observations of both hardwood and shrub height show the mean height of these groups was less within the 1-3 year fire temporal interval and greater within the midrange (4-10 year) and extended (11+) fire events. While the loss of abundance and taller cohorts of shrubs and hardwoods within burned plots is demonstrated in a variety of studies, the data indicates how quickly these can become re-established.

Hardwoods and shrubs are an important component of the biodiversity of the MPR. Some hardwoods and many species of shrubs can carry intense fires and have shown in the data they will become readily established following long term exclusion of fire. Maintaining a strong presence of hardwoods in the uplands of the MPR will conflict with the pine timber production, while silviculturally selecting against hardwoods may eventually lower the biodiversity of the ecosystem. For instance, the degree of oak (*Quercus* spp.) presence may be a function of local site conditions, as several 'oak groves' were observed in lower elevations.

Within thinned stands, it was observed that hardwoods were selected against in favor of the pine component. In the southern pine forests of the United States, silviculturalists often advise that a hardwood component should be maintained where appropriate within

stands of pine to mitigate the spread of the southern pine beetle (*Dendroctonus frontalis*) (Billings, et al., 2004). Hardwoods often support predators of the SPB, however, the specific predators of SPB and the plants which host them within the MPR ecosystem are not well known.

The ground layer vegetation, which included grasses, ferns, and small shrubs, once comprised the greatest proportion of biomass within the MPR savannah (Means, 1997). Grasses in particular represent an important part of the ground layer vegetation as they are the most abundant component. Therefore, grasses were the primary fuel in the historic MPR ecosystem as they carried the low intensity fires across the pine savannah maintaining an open condition. Wolffsohn (1967) pointed out that one years' grass growth following a fire would be enough to carry a fire in 'high' fire conditions. This was supported by the data in that no significant variation was detected in the abundance of grasses between fire temporal intervals.

Available light seems to be the most obvious limiting factor in terms abundance of grasses. However, the tolerance of this important fuel to percent foliar canopy coverage has not been quantified within the MPR prior to this study. Measurements of canopy coverage show that the majority of grass abundance was arrested when canopy coverage exceeded sixty percent. Sites such as these included areas where dense pine regeneration had been recruited following the SPB outbreak, primarily in the midrange (4-10 year) fire event sites. The dense canopies of these sites may have been further augmented by hardwoods as well as thickets of tiger fern.

Within these sites of dense canopy coverage, a high abundance of litter such as pine needles has replaced the grass component. The loss of the grass component can lead to soil erosion following intense fires in which the canopy is reopened in a stand replacement scenario.

One common vegetative fuel, dumb cane (*Tripsacum latifolium*), was not observed within any plots. Dumb cane often occurs in lower slope positions where it is assumed soil moisture levels are higher. However, in extended dry seasons, dumb cane become desiccated and has been observed carrying intense fires. Plots in this study were placed on summit positions in order to alleviate the effect of confounding variables and to observe general trends of fire temporal intervals and canopy coverage on vegetative fuels. Future studies, which include the effect of aspect and slope position on the occurrence of this fuel, may provide useable information in terms of fire management.

Pine Regeneration Response

The greatest abundance of both seedling and pole size pine regeneration fell within the midrange (4-10 year) fire event sites. This can be attributed to the SPB outbreak which created ideal conditions for seedling establishment. Pine stands under stress from the SPB outbreak may have produced excess seed which may have fallen on bare mineral soil which was recently exposed by the fire. In effect the reason midrange (4-10 year) fire events may have supported abundant pine may have been a combination of stressed pines and optimum site conditions created by fire.

With the SPB outbreak in mind, future studies will need to evaluate pine stands which have reached a dynamic climax state to accurately identify the effect of fire temporal intervals on pine regeneration.

Bare Mineral Soil and Litter

Data from fire interval measurements indicate soil exposure is limited to 4 years given re-colonization by ground layer vegetation occurs. The degree of bare mineral soil exposure following fires may be a function of fire intensity; however additional research is needed to confirm this assumption.

Exposure of bare mineral soil historically occurred following low intensity fires through the ground vegetation; however the time it remained exposed was limited as the historic grass and sedge component would have been rapidly recruited. For regeneration of pine, seed needs to be in contact with bare mineral soil for germination. Whereas if bare mineral soil is exposed for longer periods of time, such as following an intense fire, tiger fern (*Dicranopteris pectinata*) may become established on the site and may preclude pine seedling establishment.

The abundance of litter was generally above 80 percent average coverage of a plot in both the periodic and extended fire intervals. However in the recent fire events the abundance of litter was more variable between plots. In the annual fire temporal interval litter abundances ranged from 2 to 80 percent. Also as expected the degree of litter abundance was correlated with the percent foliar canopy coverage. Given the greater canopy coverage more litter will be contributed to the forest floor. In some plots, where

pine stands were dense, depth of pine litter exceeded 10cm. In sites where fire has been eliminated, litter depth will most likely preclude any establishment of pine.

Sites where bare mineral soil was exposed following intense fires should be monitored for recolonization by vegetation. Soil properties, slope position, and past management may affect the degree of vegetative recruitment, however additional research is needed.

Down Woody Debris

Measurements of down woody debris abundance relative to time since fire did not yield any significant variation between fine, coarse, and rotten woody debris. In terms of fire frequency, coarse woody debris appears to be lower in abundance following more frequent fires; however the relationship was not statistically significant. Given the humid climate of Belize, the decay rate of *P. caribaea* can be assumed to be more rapid than the decay rate of *Pinus* woods in more temperate climates of North America. The probability of the high decomposition rate may limit the degree to which down woody debris accumulates and contributes to fuel loads and fire intensity. It should be noted that fire frequency history was only measured using a 12 year fire history. Fire frequency may have a significant effect on variables if the frequency is increased and the greater temporal scale is observed.

The most pressing woody debris management issue is managing slash from thinning operations, especially in the dense pine stands, which developed following the SPB outbreak. The current priority is to thin these overstocked stands to augment forest health and facilitate timber production. The degree of thinning slash contribution to fire intensity in the MPR is unknown. These sites should be observed following treatment to

assess possible variables such as fuel loading, fuel persistence, vegetative colonization, and fire effects in thinned stands where prescribed fire has been applied.

Tiger Fern

Of all the physiognomic vegetative groups, tiger fern (*Dicranopteris pectinata*) presents one of the most challenging aspects of fuel management in the MPR. Thickets of tiger fern have been observed to carrying intense fires, especially towards the end of the dry season when the fern is most desiccated

Quantifying the abundance of tiger fern across the landscape through systematic plot sampling in the field can be problematic due to the sporadic nature of fern colonies. Ferns can be observed from aerial imagery growing in circular colonies which coalesce into larger colonies as time progresses.

A denser forest canopy may support denser thickets of tiger fern as suggested by the data. Midrange (4-10 year) fire event sites supported significantly higher abundances of tiger fern. However it should be noted these sites supported a significantly greater density of pine regeneration due to the conditions created by the SPB outbreak and following fires. The high density of these pine stands supported greater canopy coverage which may have been the governing factor for fern abundance.

Tiger fern will become readily established on sites where anthropogenic disturbance is most severe (Slocum, Aide, Zimmerman, & Navarro, 2006). In dense stands of pine, tiger fern was observed reaching heights of 2 meters indicating a high potential for it to act as a ladder fuel in which it would threaten crowns of younger cohorts of pine. Walker and Boneta (1995) found in Costa Rica that tiger fern rhizomes did not survive burning

but recovered rapidly by lateral growth from adjacent unburned thickets. The fact that sites are colonized rapidly by lateral growth would indicate small scale prescribed fires may not keep tiger fern abundances low within burn units. Tiger fern may have a greater chance of establishing on sites burned by intense fires, even into the riparian forests which experience longer term fire intervals than the upper slope positions. These fern communities will burn more readily than the hardwoods leading to increased incidence of fire within sites (Hutchinson, 1976).

The degree of tiger fern colonization may be governed mostly by the condition of the soil. Means (1997) mentions that the fern is typically found in moist settings such as in the riparian and slope transition forest, disturbed areas, or on rocky substrates. When observing satellite images of the region, it becomes apparent that tiger fern is prodigious in riparian corridors (Figure 32). There is no relationship between aspect and the relative abundance of tiger fern. However, based on extensive field observations, position on the slope may affect fern abundance, as dense tiger fern thickets are usually observed in GIS imagery inhabiting riparian zones, which supports Mean's (1997) assertion that riparian zones originally harbored tiger fern. There is no distinct explanation for the colonization of fern in upland settings.

No study has addressed the effect of anthropogenic disturbance, chiefly through logging activities, on the degree of colonization of sites by tiger fern. Heavy equipment may increase the bulk density of soils which may predispose sites to colonization by pioneer species such as tiger fern. No study as observed the effect of soil properties such as bulk density, pH, or soil moisture on the presence of tiger fern.

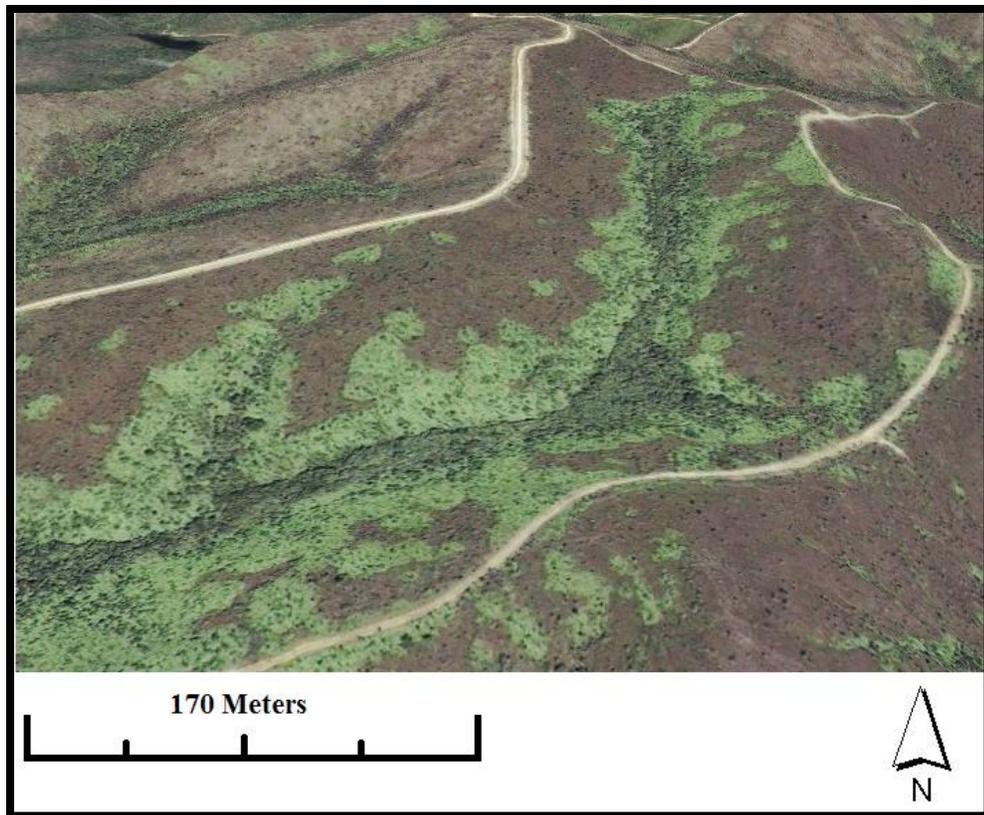


Figure 32. Typical tiger fern location on the landscape. While tiger fern may occur in scattered locations in an upland setting, it will thrive in lower slope positions such as riparian zones.

Remaining Questions

This study represented a snapshot of the effect of time since fire and percent foliar canopy coverage on vegetation, down woody debris, soil and litter. The study of fire ecology is a highly complex field in which several variables can contribute to an outcome (Whelan, 1995). The following sections will highlight some of these variables which were not observed and would be beneficial to investigate in future fire ecology studies in the region.

Fire Behavior and Intensity

Fire behavior, observed during fire events, is important in the study of fire ecology as well as in fire prescription and suppression activities (Kolaks 2004). Romme et al. (1998) point out that the response of ecosystems is a function of the severity of the disturbance. The intensity of fires is influenced by a range of factors including climate, topography (slope and aspect), fuel load, fuel type and chemistry, and the vertical and horizontal distributions of fuel (Whelan, 1995).

In this study, the effect of fire intensity on the vegetative response was not investigated. Given these fires were observed ex post facto precluded the majority of these variables being measured, including fuel load, fuel type and chemistry, fuel distribution, and weather factors. However fires examined ex post facto could have taken into account slope and aspect when vegetation parameters were measured. Future studies could incorporate the effect of slope and aspect given the installation of additional plots which can be justified to accurately sample the vegetative response relative to slope position.

A variety of studies have examined fire intensity during and recently following fire events. These studies often utilize variables such as flame height to estimate the intensity of fires and correlate the intensity to the ecosystem response. In a study assessing the effect of fuel loads on fire intensity in the Missouri Ozarks, Kolaks (2004) utilized passive flame height sensors, in which fire retardant string was placed within burn units prior to ignition to measure fire intensity. A variety of studies also utilize proxies such as char height on the bole of trees to infer fire intensity. Stevenson (2007) utilized char

height on upland species such as oaks to determine the effect of prescribed fires on Missouri Ozark forests.

Similar studies could be conducted within the MPR, however such studies may be more difficult given the ecosystem is fire dependent, making such controlled experiments difficult. Such a study would most likely be conducted during controllable circumstances, avoiding the height of the dry season when fires typically occur. For example, available fuels such as down woody debris and the vegetation could be examined prior to ignition and the char heights on the boles of pines within the fire unit can be examined post facto.

Seasonality of Fire

Seasonality of fire affects the intensity at which fires will burn in a variety of ecosystems (Kolaks, 2004). In the MPR high intensity fires generally occur during the latter half of the dry season (April through May) (Means, 1997). To avoid the potential of high intensity wildfires that occur in fuel types which historically did not occupy the MPR (such as dense colonies of *D. pectinata*), managers typically conduct prescribed burns during the early dry season or whenever fuel moisture conditions allow for a low intensity fire. The effect of these early season as well as higher intensity dry season burns on the ecosystem should be evaluated. The data from such a study could determine if a significant difference exists between burn seasonality and variables such as pine mortality and soil loss between burn units.

Considerations and limitations of Fire Ecology Research

Forest fires are the most common disturbance on earth and vital for a variety of terrestrial ecosystems (Rodriguez & Fule', 2003). However, ecological studies of fire are often lacking in planning and implementation, failing to correctly ascertain the answers for many ecological questions related to fire. The difficulty many ecologists face when studying fire is due in part to the ambition of ecological studies to measure the effects of a highly complex phenomenon that is governed by a variety of variables. Variables that may have affected fire intensity and the subsequent vegetation response in the MPR for example included humidity, ambient temperature, wind speed and direction, fuel moisture content, fuel compactness, and fuel loading. None of these variables were recorded; therefore it is not possible to completely understand the effect of the ecosystem response following fires at the present time; however with further monitoring these data may be available.

One of the most pressing issues in fire ecology is the problem of pseudoreplication. Hulbert (1984) brought this problem to light by critiquing ecological studies which he believed showed pseudoreplication. Hulbert defines pseudoreplication as the use of inferential statistics to test for treatment effects with data from experiments where either treatments are not replicated (though samples may be) or replicates are not statistically significant. In terms of fire ecology, the classic case of pseudoreplication is inferring results from just one burned and unburned treatment with several samples within each of these treatments treated as independent units.

One way ecologists overcome the problem of pseudoreplication is by conducting highly controlled experiments. Mantgem et al. (2001) delineates fire studies into three broad categories based on the amount of planned intervention in the application of fire: experimental fires, managed (or prescribed fire) and wildfires. Mantgem (2001) explained that both experimental and prescribed fires make for easier design and application of an experimental study. Conducting highly controlled experiments prove difficult in fire dependent ecosystems, especially where fire management is currently in an embryonic stage such as in the MPR.

Whelan (1995) pointed out that an ideal experimental approach, with appropriate replication can be difficult to implement within a fire prone environment. Installing and maintaining experimental blocks of treatment can be difficult. Weather conditions dictate when a prescribed burn for experimental blocks can be conducted, so some years may be too dangerous to conduct fires in fire prone ecosystems while other years may be too wet for ignition. Keeping unwanted wildfires out of experimental plots becomes more difficult with every year in a fire dependent ecosystem. Finally, Whelan (1995) indicates the problem of scale in such studies. Fires conducted on a smaller scale such as experimental blocks less than a hectare may not burn with the same intensity as naturally occurring fires.

A time-series design can reveal effects of the fire, and can be measured over time instead of during one field season such as this study. Time series designs are common in fire studies due to the ease of data collection and analysis (Whelan, 1995). The influence of fire can be long-lasting, making ecological studies time consuming and costly (Whelan, 1995). A solution to this problem is to conduct a synchronic study which is also referred

to as a 'space-for-time-substitution' (Pickett, 1989). This was the approach that was implemented in this study. However it should be noted that there are serious limitations to the synchronic approach (Whelan, 1995). The main problem is potential covariation of fire history differences and observed population differences with a third unknown site-specific variable. For example, a difference in soil type may support a different suite of vegetation affecting the relative abundance of available fuels. To overcome this, attempts were made to limit covariation such as slope aspect and logging history.

Considerations and Constraints of the Study

It is important to note that this study attempted to evaluate broad causes for this ecosystems response to fire and canopy coverage and to understand that fire ecology is a highly complex field of study.

Conducting highly controlled experiments in fire dependent ecosystems such as the MPR presents a difficult task. Restricting wildfire from any experimental block over time would be difficult or if not impossible given the lack of resources in the region. Since several fire events had been mapped since the late 1990's simultaneous measurements occurring in sites of varying times since fire, a chronosequence of fire events could be constructed. Due to the time and budget constraints, all sites were measured within one field season to construct a natural snapshot experiment or NSE. Diamond and Case (1986) describe a NSE as a condition where several sites are measured while the variable (in this case fire) is present at different levels.

One problem with using multiple fire events of varying locations is covariation, in which an unknown factor is affecting the variable of interest. In this study the soil type was a

factor which was not well understood and may have contributed some error. However fire may have greater effect on the system than the soil which supports the vegetation.

It is important to note that plots were sampled within larger fire units, assuming the plots were independent in terms of the degree of disturbance which they experienced from fire. This may be a source of pseudoreplication; however no plots were installed within 100 meters of another plot. This was done to increase the sample size which was limited due to the fact that the scale of the fire units was quite large. Mantgem (2001) mentions the fact that samples are not properly replicated does not invalidate interesting findings. However, Mantgem (2001) mentions that pseudoreplication should be mentioned where it may occur.

Fire Mediated Succession at the Mountain Pine Ridge

The Nature Conservancy (2002) developed a fire regime model based on general observations of pine savannah response to varying fire regimes (Figure 10). Following this study a new fire regime model for the MPR (Figure 33) was developed based on data and observations. The arrow on the image represents a time since fire continuum with the left portion indicating annual fires and the right portion indicating a condition in which fire is eliminated from the ecosystem. Below the continuum ecosystems are strategically located based on the typical ecosystem response to position on the time since fire continuum. The effect of intense fires as well as logging on these ecosystems is also represented; both events which may predispose sites to changes in the condition of the soil leading to invasion by tiger fern.

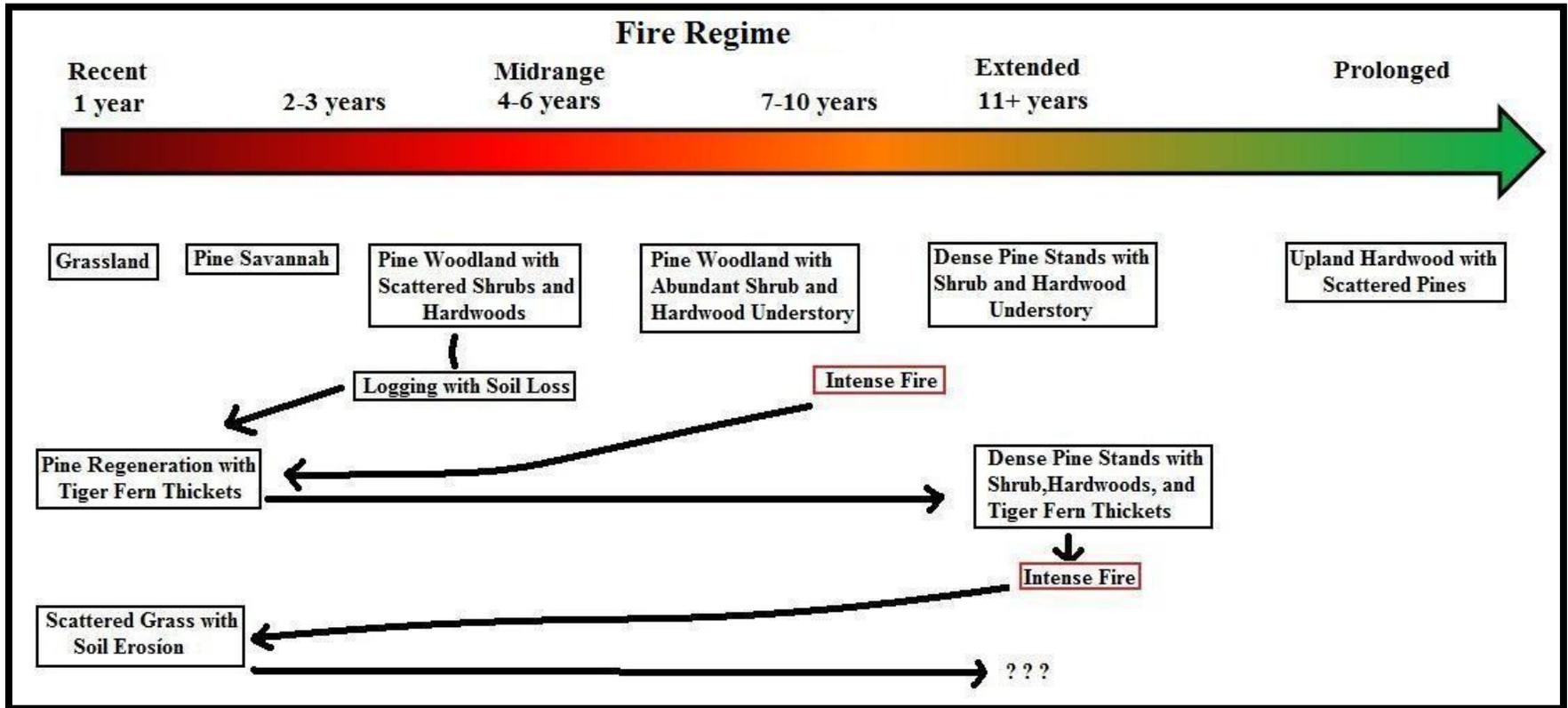


Figure 33. Conceptual fire regime model and plant community development based on data and observations from the MPR

Major Conclusions

1. Sites with more recent fire events had lower abundance of hardwood and shrub physiognomic groups and a greater abundance of bare mineral soil.
2. Fire frequency, following the SPB outbreak, did not significantly affect vegetative fuel, down woody debris, or bare mineral soil.
3. Pine regeneration response following the SPB outbreak indicates the midrange (4-10 year) fire event sites support the greatest abundance of pine, possibly owing to seed production and seed bed conditions created by the outbreak.
4. Increasing foliar canopy coverage positively affected fern and negatively affected grass vegetative physiognomic groups.
5. No significant relationship existed between tiger fern (*Dicranopteris pectinata*) abundance and aspect.

The restoration of the MPR ecosystem as well as the future of the timber management in the MPR will depend on research similar to studies such as this and implementation of practices based upon it. Understanding how to balance biodiversity and sustain the utilization of timber resources in the MPR will be an ever-evolving process. This study indicated how time since fire and percent foliar canopy coverage could affect management of fuels in the MPR ecosystem.

Dense stands of pine regeneration are forming in some sites following the SPB outbreak, especially in sites that experienced fire in the months preceding the SPB outbreak.

Silvicultural practices such as thinning and prescribed fire are currently being implemented and need to be continued to promote the sustainability and restoration of

this ecosystem. The recovery of this ecosystem depends on the continued involvement of dedicated stewards of the land as well as the support of outside sources. With continued research and its application the expedited restoration of this unique ecosystem and the recovery of sustainable resource utilization are promising.

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