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Preface

The 1999 Symposium "Fire Management: Emergent Policies and New Paradigms" was the third of an annual series of conferences presented by the California Association for Fire Ecology (CAFE) and several state and federal agencies in cooperation with University Extension, University of California, Davis. This symposium was to dedicated to fire ecology and fire management with the plenary session focusing on the integration of research, policy and management activities. Other sessions focused on a variety of fire ecology and fire management topics. This conference served as an important gathering, emphasizing interaction between landowners, land and resource managers, local decision makers, agency scientists, academic researchers, fire prevention and protections specialists, consultants, environmental organizations, community organizations, students and others interested in fire issues.

Human relationships with fire have been a major part of their interactions with natural ecosystems for millennia. Native Americans greatly influenced fire throughout the western United States, and their removal had great effects on many ecosystems. When European explorers first touched the shores of California, their activities, shaped by their needs and values, have changed the state's fire regimes. Formal fire policy since European settlement is a response to society's and institutions view of fire and changes as human relationships with the land, natural resources, and with fire change. Our understanding of the historical relationships between fire and society is greatly enhanced if we review the setting in which that society existed. It is common for us to blame our current fire situation on the shortcomings and lack of perspective of the past land managers. But this is rarely the case, because the needs and values of society have been the driving force of the past policies and those values have changed and will continue to change.

Our challenge today is to develop fire policies and management actions that recognize the need for both fire suppression and the management of fire as an ecosystem process and hazard reduction tool. Fire will continue as an important agent of change in many western ecosystems but we must strive to produce conditions where fire can become a positive force in most of California. This is a challenge of magnitude and complexity that is unprecedented in the history of wildland resource management. The stakes are extremely high and the future of both the relationship of wildlands to society and the integrity of natural ecosystems is at risk. This conference has served to improve our awareness of fire policy and how it is changing, and to stimulate discussion on future direction.

Neil G. Sugihara

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Sandra Cooper for her endless energy and enthusiasm, for keeping us all in line and for taking care of all of the symposium business.

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Fire Management: Emerging Policies and New Paradigms

A Policy Perspective of Wildland Fire Management

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Abstract

During the 20th century, wildland fire management policy has evolved from a strongly focused one-dimensional program of fire control to one that now recognizes the potential value of fire for resource benefits as well as the need for aggressive suppression. The specific process of identifying the need, developing a new policy, and gaining acceptance and endorsement can be time consuming and fraught with roadblocks and opposition. The path of wildland fire management policy evolution has been neither a direct one nor an easily adapted one. Policy changes have primarily occurred in response to the occurrence of significant events rather than in response to lessons learned, new scientific information, or advances in technology. Challenges, choices, and opportunities presented by wildland fire management are ever increasing and sound, clear, flexible policy is prerequisite to efficient management. Inflexibility in policy has been pervasive to fire management throughout the 20th century but effective policy of the future must be proactive and integrate learning, science, and technology on a continual basis.

Introduction

Albert Einstein once stated, “the significant problems we face today cannot be solved by the same level of thinking that created them.” As with many of Einstein’s statements, this has a far reaching meaning subject to varied interpretations. Basically, Einstein means that today’s problem solvers should not be yesterday’s problem creators. Generally, those responsible for current problems lack the ability to adequately plan for or anticipate all long-range developments. If they are charged with resolution, they will likely respond in a reactive fashion based on past experience which does not consider emerging advances in knowledge or technology. Einstein’s statement implies that in an ever increasingly complex world, problem solvers must be prepared to deal with developments in an unorthodox fashion, to embrace thinking “outside the box,” and accept that complex situations are dynamic and subject to constant flux, growth, or change. Unforeseen developments, although usually not predictable, must be expected. Decision makers must respond with solutions that gives results commensurate with needs. In other words, change is inevitable, and therefore, must be anticipated, and endorsed in order to resolve significant issues over time.

Wildland Fire Management Policy

In general, policy is defined as any governing principle or course of action. Specifically wildland fire management policy represents the source of the bounds and overall direction for fire management actions. Wildland fire management became an organized function with definable objectives around the end of the 19th century. In the following 100 years, policy has driven the development and direction of operational wildland fire management.

Wildland fire management policy establishes principles that reflect considerations of:

- land and resource management needs,
- changing fuel complexes and effects on fire behavior,
- firefighter and public safety,
- increasing complexity in land use,
- changing societal values and perceptions,
- growing technological capabilities, and
- expanding scientific knowledge of the natural role of fire and fire effects.

Many of these categories are now far more defined and developed than they were 100 years ago. Given these circumstances, Einstein's statement is applicable to wildland fire management and it's associated policy. Today's problem solvers must create dynamic and flexible policy that can adapt and demonstrate responsiveness to changes. However, during the course of organized wildland fire management, its policy makers have not been willing to adapt and embrace change as indicated by important baseline considerations. Over the 20th century, wildland fire management policy has not evolved gradually to correspond to state-of-the-knowledge growth and technological improvements. In stead it has emerged in discrete increments in response to significant events responsible for serious and adverse outcomes.

Processes of Policy Development

Policy for any functional activity must be based on certain cornerstone criteria. Wildland fire management policy is formulated on the basis of such factors as social, political, economic, environmental, and safety considerations; general land management goals; and science and technology (Figure 1). When policy is developed from consideration of these factors, it can be effective in providing a framework for operational management actions.

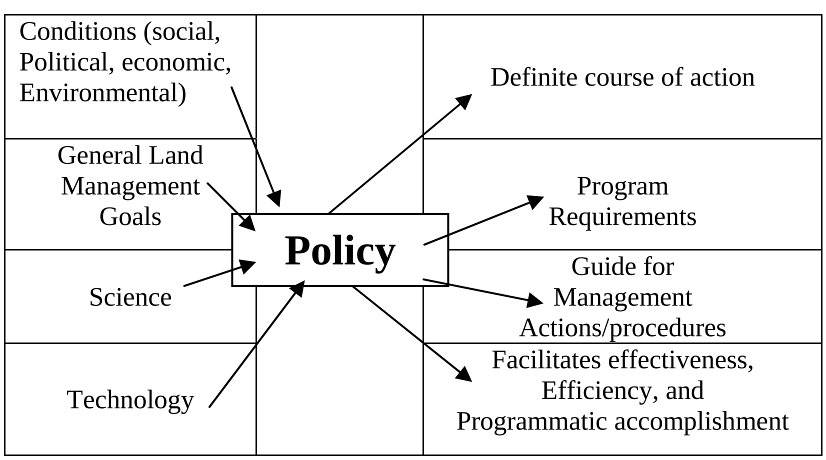

Figure 1.Policy criteria

Changes in knowledge, advances in technology, lessons learned from past experience, and changing situations all influence what policy must be. The development of wildland fire management policy has been primarily reactive in nature. The ability and opportunities to be proactive are limited by the dynamic nature of the activity and past efforts to establish and maintain fixed, inflexible standards.

A simple model of management dynamics (Figure 2) can be used to illustrate general policy formulation. Once policy has been established, management activities function in an efficient and effective way, given periodic maintenance and revision in response to the situation dynamics. But, in the absence of periodic maintenance and revision, management efficiency begins to erode over time. As shown on the right side of the model (Figure 2), diminishing effectiveness leads to a cascade of stages that can make the situation progressively worse. The first stage that develops as a result of chronic loss of efficiency can be classed as situational blindness. This is the period where a problem exists that degrades or hinders efficiency but involved parties are unaware of it. In the absence of any action, the situation progresses to the stage of passive awareness. In this stage the problem is now clearly apparent but involved parties either do not want to act on it or do not know what to do in response to it. Continued lack of action leads to a worsening of the situation until reaching the stage of active awareness. In this stage, involved parties are fully aware of the problem and have a clear majority agreement that something must be done to resolve it. The two stages of awareness, passive and active, can occur rapidly as a result of the occurrence of an acute event as opposed to a long-term chronic pathway. This type of an event is of such immediate and serious ramifications that the problem is immediately apparent and swift action is necessary.

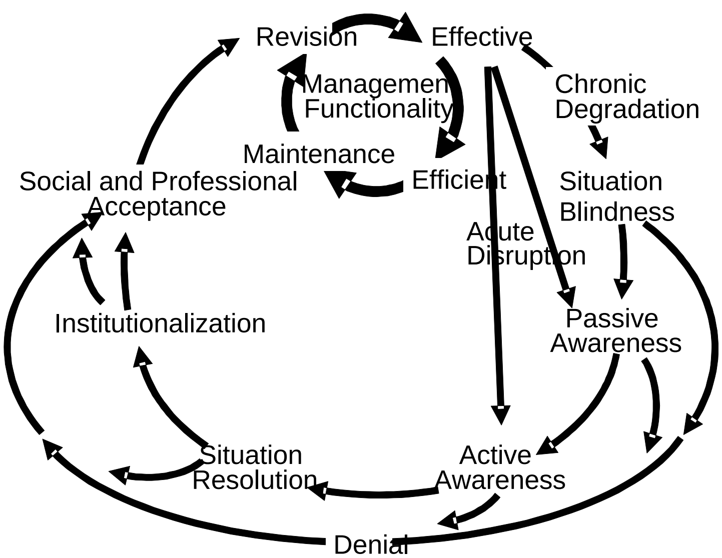

Figure 2. Management dynamics model

Once the problem has been addressed in the active awareness stage, situation resolution can be reached. In this stage, a solution to the problem has been determined. Definition of and agreement of the solution moves the situation into the institutionalization stage where new policy, procedures, and practices are derived. After institutionalization occurs, the policy (problem solution) must attain social and professional acceptance to continue to move forward.

Without both internal and external acceptance and endorsement of the new policy, it has scant opportunities for success. Social and professional acceptance can only be achieved through open communication with concerned publics and agency personnel. Once acceptance is achieved, the system is on track to return to its highest level of functional efficiency, as shown at the top of Figure 2.

At any point along the various steps in this process of policy development, another stage or situation can develop. This is the denial stage where some of the involved parties refuse to accept that there is a problem, that action needs to be taken, that any resolution that is reached is necessary, or that the solution is the correct one. Persons in the state of denial generally, through intentional or occasionally unintentional actions, delay the return to highly efficient management functioning due to reasons ranging from strong personal commitments to simple resistance to change. Informing and educating both the public and employees in regard to the new policy is necessary for the elimination of denial. Information and education are vital to full social and professional acceptance (end of the denial stage).

Once social and professional acceptance occurs, the new policy and procedures can move back into a highly efficient mode, subject to periodic maintenance and revision. When management operations are functioning effectively again, it can be expected that the cycle shown in Figure 2 will repeat itself on a variable frequency dependent upon internal and external influences.

Programmatic Development of Wildland Fire Management

Wildland fire management shows relatively slow policy growth and change throughout its history. Pyne (1982) describes wildland fire management policy development in terms of four problem fire types and management response to them: frontier fire, backcountry fire, mass fire, and wilderness fire. He classes these fire types temporally as frontier fire (1910 – 1930), backcountry fire (1930 – 1949), mass fire (1940 – 1970), and wilderness fire (1970 – present).

For the purposes of this paper, fire policy development is categorized by its interrelationships with significant event response, management actions, and subsequent technological development. Tables 1 and 2 illustrate fire policy development in this context and show during the 20th century, policy emphasis changed incrementally. These tables also show how policy changes resulted from, more often than not, the occurrence of an acute event and lacked dynamism. Periods of significant policy change are categorized as 1910 - 1940, 1941 – 1970, 1971 – 1988, 1989 – 1994, and 1995 – present (Tables 1, 2).

Initial wildland fire management policy developed in relation to pure protection objectives. Managers desired to eliminate large-scale destruction by uncontrolled wildfires of both personal property, entire towns, and the natural forest. Fire was viewed as menace to safety and to wildlands and efforts were concentrated on eliminating it.

As this focus continued and strengthened, it fueled programmatic growth of the fire control portion of fire management. Developmental steps of most significance occurred around every 20 –30 years starting about 1910 and lasted for variable lengths of time. Fire control objectives and development of tactics and equipment were most important until 1970. After 1975 the programmatic growth of fire suppression slowed, although never ceasing to expand. Fire use, on the other hand, did not play a prominent role in programmatic development until the late 1960’s. This portion of the program began prior to 1920 in various areas but represented a highly contentious function and received application only on a small scale site-specific basis.

This highly contentious nature translates into the management dynamics stages of situational blindness and denial as shown in Figure 2. In fact, with shifts in leadership in natural resource management agencies, the policy of using fire was suspended and reinstated several times from 1920 – 1940. After 1960, management ignited prescribed fire became more accepted and a name change for the program from fire control to fire management was endorsed in 1974. Following this, the fire use portion of the program began a period of markedly enhanced programmatic development.

All aspects of wildland fire management are continuing to develop and expand and expectations are for the development of individual program components to merge within the next 25 years. This means that no longer will growth and development support the fire suppression or fire use portions of the program individually, but new developments and changes will be supportive of all aspects of fire management.

Technological Advances, and Significant Milestones Associated with Wildland Fire Management Policy Development

The onset of organized fire control (around 1900) initiated a period of close attention to fire suppression strategy and tactic development (Table 1). This period continued until around 1970 with all major efforts resulting in better use of firefighters, development of equipment, application of mechanized equipment (Table 1). After 1940, efforts also began to focus on fire

Table 1. General chronological development of wildland fire management policy and important milestones, 1900 - 1970.

Significant Events Influencing Wildland Fire Management	**Event Type (chronic or acute)**	**General Policy Direction**	**General Time Period**	**Wildland Fire Management Operational and Technological Milestones**
Peshtigo wildfires, Northern Idaho/Montana wildfires	Acute	Full suppression, maximum protection. Development of fire control policy and use of crews in suppression.	1900 – 1940	Established suppression capability, Fire Detection, Hand Tool development, Crew organization, Surface access – Civilian Conservation Corps
Increased knowledge of fire and fuels	Chronic	Full suppression, maximum protection, limited site-specific prescribed burning. Continuation of fire control policy with advances in crew use and mechanization.	1941 – 1970	Use of fixed wing aircraft; Heavy equipment, bulldozers, tractor-plows; Smokejumpers; Smokey Bear Prevention Program; Advancement in tools, equipment; Training and organization; Helicopter use; Hotshot crews; Fire shirt (protection clothing); Single engine retardant aircraft; Site-specific prescribed burning; Initial drought index ; Simple model for predicting characteristics of a large stationary mass fire

prevention and initiated one of the most successful advertisement campaigns in this country, the Smokey Bear fire prevention program. Policy direction during this period was directly designed to support and enhance fire control. Resulting milestones primarily reflect operational over technological gains.

After 1970, the program evolved to recognize fire use in the form of both prescribed fire and management of naturally ignited fires as important program components in addition to wildfire suppression (Table 2). A name change occurred to accomodate this phase of program development. Technological milestones began to equal and surpass operational milestones at this time (Table 2). Many technological milestones were representative of the enhanced growth of the prescribed fire portion of the program. Application of scientific information became an accepted part of developing operational strategies and tactics and interagency cooperation became a standard practice.

Table 2. General chronological development of wildland fire management policy and important milestones, 1971 – 1988.

Significant Events Influencing Wildland Fire Management	**Event Type (chronic or acute)**	**General Policy Direction**	**General Time Period**	**Wildland Fire Management Operational and Technological Milestones**
Increased knowledge of fire ecology, the natural role of fire, fire and fuel dynamics, and advocacy of prescribed fire and management of naturally ignited fires for beneficial purposes	Chronic	Name change from fire control to fire management, expansion of prescribed fire program and management of naturally ignited fires for beneficial purposes. Policy beginning to recognize other aspects of fire management.	1971 – 1988	Policy change from Fire Control to Fire Management to expand programmatic functions to include prescribed fire , Fire suppression overhead plus Fire Behavior Officer position, Fire qualifications system, Personal protective equipment (pants, shirts, shelter, hoods), Multi-engine airtankers, Aerial ignition, Greater use of prescribed fire and initial use of prescribed natural fire, Intensive training program, Fire behavior prediction, fire danger rating, Formal fire planning, Fuel moisture monitoring, Incident Command System, Large helicopter, Improved crew program

The 1988 fire season signaled the start of the most intense period of programmatic growth and in federal land managers' ability to accomplish wildland fire management objectives. Since 1988 (Table 3), a cascade of events has been instrumental in a technological development

Table 3. General chronological development of wildland fire management policy and important milestones, 1989 – present.

Significant Events Influencing Wildland Fire Management	Event Type (chronic or acute)	General Policy Direction	General Time Period	Wildland Fire Management Operational and Technological Milestones
1988 Fire Season (Yellowstone Fires) – Federal Fire Policy Review	Acute	Greater long-range accountability for fire management, full program of suppression, prescribed natural fire, and prescribed fire. Beginning of shift from suppression paradigm to prescribed fire (including management and naturally ignited fires).	1989 – 1994	Began the most intensive period of programmatic development, technology development, research applications, and policy and program review One policy review (1989) One General Accounting Office Audit (1990) Formal prescribed fire curriculum and qualifications system Long-term accountability (regional and national preparedness levels) Extensive Fire Management Planning Greater attention to safety
1994 Fire Season – South Canyon Fire Review , Interagency Management Review Team Report, Federal Fire Management Policy and Program Review	Acute	Wildland Fire Management, greater attention on safety, increased emphasis on managing wildland fire for resource benefits, additional funding to establish and implement an aggressive program of hazardous fuel treatment. Full shift from suppression as leading fire management action to fire suppression and use paradigm. Application of appropriate management response to all wildland fires allows managers greater flexibility and efficiency in fire management. Greater fireline and public safety and greater attention to fire management expenditures.	1995 – present (2000)	One policy review (1995) One General Accounting Office Audit (1999) Greater information acquisition, analysis, application, and archival capabilities with computers Geographic Information Systems, spatial fire management systems and analysis Internet, infrared and digital photography Long-term risk assessment processes (rare event analysis) Large fire growth simulation Analysis of potential fire effects Continued advances in fire behavior prediction Crown fire prediction Prescription development Smoke emissions and dispersal modeling Satellite imagery for assessing fuel conditions Better access and analysis of fire weather information Rapid access and display of fire behavior and danger products Greater attention to safety Dedicated fire use resources to facilitate both suppression and fire use Continued advances in tools, protective clothing and shelters, and other equipment Better application of mechanized equipment

explosion to support fire management decision-making and increase accomplishments in all aspects of the program. Advances in decision support tools and computer software have facilitated the ability to acquire and analyze data. Two reviews of federal wildland fire management policy and two General Accounting Office program audits have occurred that have been largely responsible for the creation of new or revised internal agency procedure guidelines, agency manuals, and improved interagency coordination standards and procedures. In aggregate, these activities have fostered forward-thinking fire management planning and encouraged more informed decision-making. The long-term fire program accountability and interagency cooperation has become the cornerstone of wildland fire management.

The Federal Wildland Fire Management Policy developed and approved in 1995 directs management of wildland fire to accomplish a full range of management objectives through application of an appropriate management response. The National Wildfire Coordinating Group developed an "umbrella" flowchart to illustrate the broad framework within which the policy will be implemented (Figure 3). The development of this policy closely mirrors the management

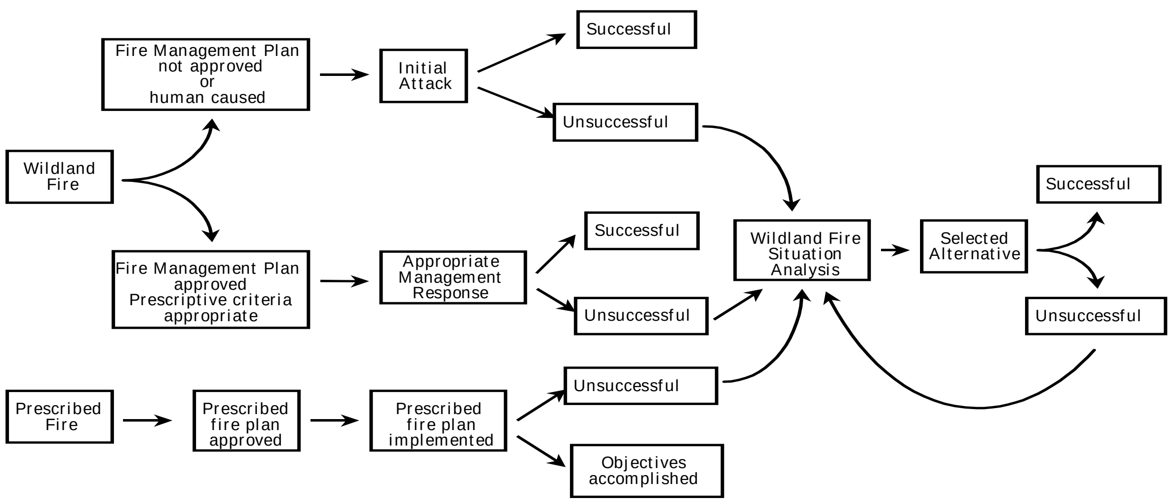

Figure 3. Umbrella flowchart

dynamics model shown in Figure 1. After the 1989 policy review (U. S. Department of Agriculture/U. S. Department of the Interior 1989) and a program audit in 1990 (U.S. General Accounting Office 1990), the natural fire policy was endorsed as sound, but very conservative implementation procedures were adopted (Agee 1997). These procedures severely limited the ability to use natural fire for beneficial purposes due to lengthy and cumbersome paperwork requirements, long-range preparedness planning levels that limited availability of staffing and equipment for these fires and advocated suspension of natural fire management during periods of high suppression activity. Other limiting factors included highly restrictive funding limits, risk aversion by many managers, and less than full internal support within agencies. These factors combined to present a conservative viewpoint that led to decisions to suppress many fires rather than manage them for resource benefits. This situation quickly led to the situation blindness stage, but before it could fully shift to passive awareness, the South Canyon Fire occurred in Colorado in 1994. This fire, unfortunately responsible for 14 firefighter fatalities, represented an acute event of such severity that wildland fire management was immediately directed into the active awareness stage.

Response to this event produced the Federal Wildland Fire Management Policy and Program Review (U.S. Department of the Interior/U.S. Department of Agriculture 1995), and preparation of new agency manuals and guidelines, and an interagency policy implementation procedures reference guide (Zimmerman and Bunnell 1998). Completion of these materials moved wildland fire management into the institutionalization stage.

The Federal Fire Policy emphasizes personal safety, intensive fire management planning, the incorporation of the best available science, and incorporation of the natural role of fire and its effects on natural ecosystems as vital components of planning activities. Under this policy, management responses are developed for each individual wildland fire. Management actions can response to resource management needs and constraints, safety, and cost while maintaining the flexibility of planned actions as conditions change. Many of the previous barriers to successful use of fire for resource benefits have been eliminated through this policy.

Ongoing education and information efforts have moved this policy revision into the social and professional acceptance stage. Completion of new fire management plans and understanding by all agency personnel will signal endorsement and movement back to the most efficient level of wildland fire management. The Federal Fire Policy represents the latest step in wildland fire management policy evolution which coupled with continued technological advances is molding the archetypical model of wildland fire management for the immediate future.

Conclusions

Policy must be dynamic because at any given time, the cutting edge issues may have been unexplored at the time of policy development. Conversely, the importance and urgency of past issues may wane over time. This dynamic nature is very well described by Lee Thomas in a statement made to a U.S. Senate Environment and Public Works Committee confirmation hearing in 1985 when he said, “there are a whole new generation of issues before us that will require a new approach to environmental management.”

Quality communication is the single facet of policy implementation most important to achieving social and professional acceptance and removing denial, critical stages to achieving and maintaining management effectiveness. All natural resource managers are familiar with the term, “adaptive management,” but policy makers must understand and promote “adaptive policy” as an important working strategy. Perpetual adaptation to changing needs and limitations is vital to long-term policy effectiveness.

During the last 100 years, wildland fire management policy has followed a slow and steady course. Development has occurred quickly in response to significant events rather than adapting to developments in important baseline considerations. The future of wildland fire management and its policy are concisely summed up by another quote from Albert Einstein, “out of clutter, find simplicity, from discord, find harmony, in the middle of difficulty lies opportunity.”

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A Risk-Based Comparison of Potential Fuel Treatment Trade-off Models

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Abstract

Understanding the trade-off between short-term and long-term consequences of fire impacts on ecosystems is needed before a comprehensive fuels management program can be implemented nationally. We are comparing three vegetation models that may be used to predict the effects of various fuel management treatments at eight locations in major U.S. fuel types. We selected the **F**ire **E**ffects **T**rade-off **M**odel (FETM), the **SIM**ulating vegetative **P**atterns and **P**rocesses at **L**andscape sca**LE**s/**M**ulti-resource **A**nalysis and **G**eographic **I**nformation **S**ystem (SIMPPLLE/MAGIS) modeling system, and the **V**egetation **D**ynamics **D**evelopment **T**ool/**T**ools for **E**xploratory **L**andscape **S**cenario **A**nalyses (VDDT/TELSA) models for testing. We are evaluating the implementation of each model and will estimate the uncertainty associated with predictions from the three models using simulation. This uncertainty is a component of the risk associated with a fuel management program. The model comparison will identify model components that are needed for a national strategic fire-planning model.

Introduction

It has long been known that silvicultural operations such as thinning and prescribed burning can modify and/or reduce fuels in many forested systems. Recent developments have heightened interest in the use of these management tools to reduce the risk of large stand-replacing wildland fires (USDI/USDA 1995, GAO 1999) as well as treating accumulated fuels in non-forest ecosystems. Increased use of prescribed fire may also have short-term consequences on air quality, recreational use, property and ecosystem structure and function. Land and fuel managers must understand the trade-off between short-term and long-term consequences of fire impacts on ecosystems before a comprehensive fuels management program can be implemented at the national level. The uncertainties associated with a fuels management program must be clearly understood and quantified. These uncertainties include undesirable ecological effects, prescribed fire escapes, decreased atmospheric visibility and degraded air quality. Lack of fuel treatment presents its own set of uncertainties, including large stand-replacing fires, abnormal ecosystem dynamics, and periods of locally heavy smoke emissions.

Mathematical models can be useful for quantifying the risks and trade-offs of fuels management policies and programs. Given the long time horizon (50 years) associated with land management planning, models are indispensable for providing managers with information on future landscapes. These models can range from simple growth and yield models to elaborate process simulation models. Regardless of type, models are only approximations of reality. As a result, their outputs are subject to differing degrees of uncertainty and error. This introduces an element of risk to decisions based on evaluations from these models.

A number of models are presently in various stages of development and application for use in understanding and predicting the effect of fuel management strategies on forest health, smoke emissions, and commercial harvest. Most models have been applied to few locations, so it is not clear if any model can be applied nationally. Documentation and evaluation of models have been sporadic, due largely to development needs being driven by regional and sub-regional needs, rather than national. An assessment of the number and scope of models available indicated that presently no model contains all of the desired abilities needed for a nationally applied model system. Several models might potentially develop into a nationally applied trade-off model, but it is not clear from literature reviews how the models actually perform in field applications.

We have been funded by the USDA/USDI Joint Fire Sciences Program to address some of these questions surrounding fuel treatment models and their use. Our objectives are:

1. To perform a comprehensive sensitivity analysis of SIMPPLLE/MAGIS, VDDT/TELSA, and FETM to determine the reliability of each model, and document the justification of the approach used in the internal algorithms;
2. To parameterize FETM, SIMPLLE/MAGIS, and VDDT/TELSA at 8 locations representative of major fuel types found on lands managed by USDA, USDI, DOD, and state agencies;
3. To simulate a set of fuel treatments for each model and compare/contrast model results with regard to wildland fire occurrence, smoke emissions and vegetation distribution; and
4. To develop methods for estimating the uncertainty (risk) associated with vegetation changes resulting from fuel treatments in each of the fuel types studied.

Although the 3 modeling systems overlap in functionality, design emphasis differs substantially among the three. SIMPLE/MAGIS is designed to provide decision support at the project/watershed level for treatment type and sequencing on the landscape (Jones et al *in press*). FETM (Schaaf and Wiitala *in press*) and VDDT/TELSA (Arbaugh et al *in press*), are larger scale planning models, intended to support forest level decisions. FETM has a more detailed prescribed and wildland fire component than VDDT to emphasize the trade-off of fire management actions. VDDT is a more general model than FETM and includes a large array of other disturbance factors presently not available in FETM.

Background and Literature Review

Quantitative fuels trade-off planning using simulation models is a recent innovation. Fuels trade-off models have been under development only since the late 1980's. These models are a subset of the many vegetation disturbance models that have developed recently (Schmoldt et al. 1999). During the 1990's several separate modeling developments were initiated or adapted to examine aspects of the general question of trade-off between fuel treatments and the effect of these treatments on wildland fire hazards. A more detailed literature review of the potential models and the process used to select the models being evaluated can be found in Weise et al. *in press*.

The terms "risk" and "hazard" are sometimes used synonymously. However, in decision-theory, risk is defined as a function describing the expected loss associated with a particular decision rule. Hazard can be defined as a potentially dangerous condition. Feary and Neuenschwander (1998) defined hazard as "a threat to humans and their welfare" and risk as "the probability of hazard occurrence." With the advent of remote-sensing imagery and geographic information systems, numerous authors have developed methodology to describe both fire hazard (potentially dangerous situations) and fire risk (probability of those situations occurring) (e.g. Chuvieco and Congalton 1989, Vidal et al. 1994). Various approaches have been or are currently being applied to state and federal lands throughout the United States. While these efforts have many common elements, they are usually tailored to meet local needs.

Risk analysis (or assessment) has been developed and applied in several different fields (Molak 1997). Some common elements of the various risk analysis processes are 1) identification of the hazard, 2) relating a quantifiable measure of the hazard to an adverse effect, 3) determining who or what is exposed to the hazard, and 4) calculating risk using 1, 2, and 3 (NAS 1983 cited in Molak 1997). Risk assessment in wildland fire has historically focused on determining the risk of fire occurrence. Regional fire danger rating systems evolved into the National Fire Danger Rating System (NFDRS) which quantifies the probability of a fire occurring within a particular management area (Deeming et al. 1977, Bradshaw et al. 1983). Remote-sensing and geographic information systems have been utilized to estimate fire risk by using satellite imagery of vegetation (e.g. Gonzalez-Alonso et al. 1998, Chuvieco and Congalton 1989). Components of the NFDRS are being coupled with satellite imagery to estimate fire potential (Burgan et al. 1998). The focus of this type of modeling is on fire occurrence modeling. With the increased availability of this technology to land managers, wildland fire risk assessment is now beginning to estimate the risk posed to various resources by a wildland fire (e.g. Burton et al. 1998). This is one of the 4 program areas of the federal Joint Fire Science Program as described in the 1998 Joint Fire Science Plan (available at URL http://www.nifc.gov/joint_fire_sci/JointFire.html or from the 1st author).

In the context of the present study, we will look at the risk associated with use of each model. In other words, we will estimate the uncertainty (or reliability) of the models using simulation and other appropriate analytical techniques. Model uncertainty can be thought of as a component of the risk involved in making a fuel management decision. Our work in the current project can not be described as a full-blown risk analysis. The hazard (wildland fire) has been sufficiently characterized. For example, information linking wildland fire to adverse effects is known for a variety of ecosystems, wildland fire exposure analysis is typically conducted locally by fire managers, and various means to calculate risk have been developed as discussed above. We will only describe the uncertainty of the model tools, not the uncertainty of the information used to initialize or parameterize the models.

Research Methods

Sites have been located in both the western and eastern U.S. in both forest and shrub ecosystems that are managed by several federal agencies (see Table 1). Current vegetation and topographic information stored in a GIS system and a comprehensive fire history database containing fire occurrence by final size and vegetation class are necessary data for all models. The models are intended for application at diverse geographical scales ranging from < 25,000 acres to > 1,250,000 acres. We will simulate fuel treatments on landscapes of 250,000 – 500,000+ acres, a common size where all models should perform.

Table 1. Location and major U.S. fuel type being used in comparison of potential fuel treatment trade-off models.

Location	Fuel Types
Yosemite NP, California	various Sierra Nevada
Angeles NF, California	chaparral
Bitterroot NF, Montana	various northern Rockies
Gila NF, New Mexico	various southern Rockies
Alabama/Florida	longleaf pine
Huron-Manistee NF, Michigan	jack pine
Utah	sagebrush, pinyon-juniper
Kenai Peninsula, Alaska	white spruce with significant insect-caused mortality

Model evaluation will begin with a comprehensive sensitivity analysis of each model to determine the present limitations of each model for national application, and any areas where internal algorithms fail. Given the complexity of these models, analytical methods of partial derivatives are impractical. Instead sensitivity analysis will be performed using simulation techniques. Key portions of the models will be examined in detail. The objective of the sensitivity analyses is to determine which of the input variables or processes have a strong influence on model predictions. As an example, the influence of flow rates and probabilities in the vegetation transition matrices of FETM and VDDT/TELSA, respectively, on final vegetation distribution can be determined. FETM is a "stock and flow" model where successional movements are handled by flow rates or transition coefficients as opposed to the probabilities used in VDDT.

Following the sensitivity analyses, two types of model evaluation will be conducted. First, a retrospective model application and second an application to test sites. We are conducting retrospective studies at two sites: Yosemite National Park and a second site to be chosen. Historical vegetation distribution for Yosemite National Park is based on the Vegetation

Type Maps (VTM) and plots that were surveyed by the National Park Service in conjunction with the California Forest and Range Experiment Station in the 1930s (Wieslander 1961). The purpose of the retrospective analysis is to determine whether models, given known historical vegetation, management and wildland fire information, can reproduce current ecosystem structure (Figure. 1). Since not all of the models are spatial, only total area in each vegetation class will be compared for all models. Spatial analysis techniques will be used for those models with spatial output. One potential measurement of agreement between observed and predicted vegetation distribution is the Kappa statistic used in the analysis of error matrices in remote sensing and classification (Congalton and Green 1999). Components of the models (such as the mechanism to simulate fire occurrence and size) will also be compared with observed data.

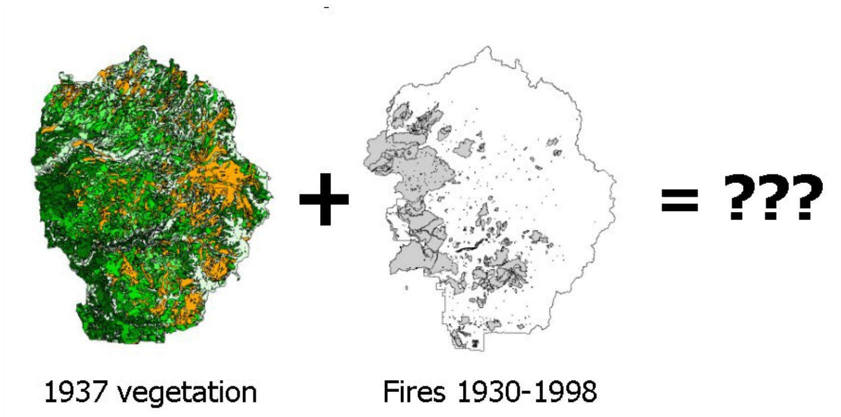

Figure 1. Schematic representation of historical retrospective analysis at Yosemite National Park. The 1937 vegetation is based on the Vegetation Type Mapping survey (Wieslander 1961).

In addition to the retrospective analysis, the models are being applied to several sites nationally to determine adequacy for present day planning and trade-off needs. Both the technical aspects needed for trade-off evaluation and the user-friendliness (ease of use) of the models will be evaluated. A principal trade-off evaluated will be smoke emissions. Emission factors and fuel consumption rates will be kept constant for each model so differences in total emissions for each model will be a function of the predicted area for each vegetation class and the proportion of the vegetation class burned. In addition to total emissions, the predicted proportion of total area occupied by each vegetation class will be compared between the individual models. The emphasis of the model comparison analysis will be 1) to test if general vegetation predictions are comparable between models, and 2) to identify future model development changes for each model that will ensure comparability of basic vegetative and management effects between models, rather than to identify a 'best' model.

Model assumptions and formulations will be examined to determine differences in model outputs for each location. Once each model is parameterized for a location, estimating the uncertainty associated with model projections is a relatively simple effort. We will estimate confidence intervals for total emission estimates as well as area occupied by each vegetation class. The process is similar to sensitivity analysis. Values of the input variables will be altered according to published values and expert knowledge. Model outputs will be summarized to produce empirical probability distributions for vegetation distribution and total emissions. Since predicted vegetation will be one of several possible types, the empirical distribution function will most likely be the multinomial distribution. The empirical frequency distributions describe the uncertainty of the model outputs and the risk associated with model errors.

Progress to Date

Data have been compiled from the initial three sites - Angeles NF (~684,858 acres), Yosemite NP (~747,000 acres) and the western portion of the Stevensville District of the Bitterroot NF (~471,293 acres) (Figure 2). These sites represent a range of management activities and disturbance regimes. The Angeles National Forest, located in the San Gabriel Mountains of southern California, is composed principally of shrub lands with a small percentage of coniferous forest and oak woodlands (Table 2). As a result, there is very little timber harvesting activity and the principal disturbance factor is fire. Vegetation of Yosemite National Park, located in the central Sierra Nevada Mountains of California, consists principally of coniferous forests of various types with a small percentage of oaks and other hardwoods (Table 3). With the exception of small areas along the western edge of the park, the principal disturbance since European settlement has been lightning-caused fire with human-ignited prescribed burns occurring since the early 1970s. In contrast to these sites, the Bitterroot

Angeles NF Bitterroot NF Yosemite NP

Figure 2. Mapped fire occurrence in Yosemite National Park from 1930 to 1998, Angeles National Forest from 1878 to 1995, and the Bitterroot National Forest from 1877 to 1998. Maps are to scale.

National Forest has experienced substantial timber harvest activities as well as fire. Insects and diseases are two other disturbance factors in the forest portions of Yosemite and the Bitterroot National Forest that are known. The principal vegetation type of the Bitterroot is mixed conifer forest with a small percentage of quaking aspen (Table 4).

While all three areas experience wildland fire, the number of fires differs appreciably between the locations. The period of recorded fires also differs appreciably. The fire size distributions presented in Figure. 3 were derived from fire maps or from fire databases maintained by the locations. The minimum fire size reported and years of record differ between

GIS coverages and databases. Regardless of the differences in the data, the size distributions are similar as would be expected with a large number of small fires and a small number of large fires.

The models are currently being parameterized for the three first year locations and sensitivity analysis is underway. We have selected the Bitterroot National Forest as the first site to implement all three models, simulate fuel treatments, and compare results. The results of this comparison will be available for presentation in September, 2000. We will begin to parameterize the models for longleaf pine and the southern Rockies in 2000 and the remaining three locations in 2001. The project is scheduled to be completed by 2002. Contact the senior author for additional information about the project.

Table 2. Vegetation coverage of the Angeles National Forest based on classified digital imagery and ground-based verification (Franklin 1998).

Vegetation type	Acreage	Vegetation type	Acreage
Northern mixed chaparral	360521	California black oak	1116
Semidesert chaparral	53086	Blue oak	1539
Chamise chaparral	61163	Riparian	1947
Mixed desert scrub	617	Canyon live oak	53041
Montane mixed chaparral	18264	Coast live oak	15142
Digger (gray) pine	709	California walnut	30
Pinon	28744	Buckwheat, white sage	38421
Mixed conifer - pine	11869	Coastal sage scrub	47957
Bigcone douglas-fir	44665	Big basin sagebrush	4753
Mixed conifer – fir	48822	Barren	7424
Jeffrey pine	5522	Urban	27707
Coulter pine	564	Water	7043
Annual grass/forb	9999	Agriculture	2156
Valley oak	408		

Table 3. Vegetation distribution for the western half of the Stevensville Ranger District, Bitterroot National Forest based on digital imagery and ground-based verification.

Vegetation Type	Acreage	Vegetation Type	Acreage
Alpine fir (AF)	29543	AL-WB-AF	3956
Alpine larch (AL)	2630	DF-AF	2364
Black cottonwood (CW)	429	DF-GF	201
Douglas-fir (DF)	52460	DF-LP	1148
Grand fir (GF)	72	DF-LP-AF	16527
Larch (L)	89	L-DF	6606
Lodgepole pine (LP)	18636	L-DF-AF	14310
Ponderosa pine (PP)	26221	L-DF-GF	235

Vegetation Type	Acreage	Vegetation Type	Acreage
Quaking aspen (QA)	455	L-DF-PP	690
Whitebark pine (WB)	7435	L-DF-PP-LP	71
CW-Mixed conifer (MC)	17852	L-LP	125
Engelmann spruce (ES)-AF	1737	L-LP-DF-AF	2342
QA-MC	946	L-LP-DF-GF	255
WB-ES-AF	23394	L-PP	697
Non-forest	193113	L-PP-LP	77
Non-soil	3399	PP-DF	39229

Table 4. Vegetation distribution in 1937 for Yosemite National Park based on maps and ground surveys produced by the Vegetation Type Mapping project (Wieslander 1961).

Dominant vegetation	Acreage	Dominant vegetation	Acreage
Lodgepole pine	165216	Mountain whitethorn	2388
Jeffrey pine	94112	Deerbrush ceanothus	322
Digger pine	2491	Wedgeleaf ceanothus	283
Ponderosa pine	86423	Littleleaf ceanothus	557
Red fir	87427	Chinquapin	1287
White fir	43350	Birchleaf mountain mahogany	615
Mountain hemlock	35392	Bitter cherry	173
Whitebark pine	35155	Sagebrush	499
Incense cedar	577	Timberline sagebrush	305
Western white pine	21910	Chamise	137
Sugar pine	19432	Cream bush	34
Western juniper	8857	Bear clover	1226
Douglas-fir	4948	Meadow	24419
Canyon live oak	22575	Herbaceous	663
Huckleberry oak	7682	Rush	260
California black oak	3581	Grass	68
Interior live oak	952	Alpine lupine	223
Brewer's oak	46	Bolander's locoweed	137
Black cottonwood	40	Lupine	20
Willow	3134	Mountain mule-ears	9
Quaking aspen	1981	Barren	115596
Mountain alder	9	Glacier	90
Big-leaf maple	61	Lake	7366
Mountain maple	6	Cultivated	32
Greenleaf manzanita	3993	Orchard	2
Pinemat manzanita	79	Unknown	1005
Mariposa manzanita	79		

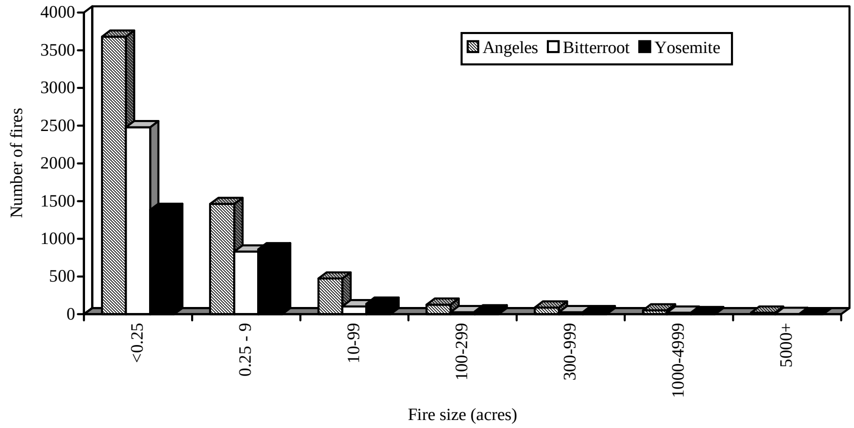

Figure 3. Fire size (in acres) distributions for Yosemite National Park (YNP), Angeles National Forest (ANF), and Bitterroot National Forest (BNF). The data source for YNP is the fire occurrence map in Fig. 2. The data sources for ANF and BNF are PCHA fire occurrence databases. Period of record for YNP is 1930-1998, ANF, 1911-1998, and BNF, 1970-1998.

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Design, Implementation and Evaluation of a Multi-Scale Prescribed Burning Program on Santa Cruz Island.

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Abstract

As part of multi-island conservation management program, The Nature Conservancy and National Park Service designed and implemented a series of research and management burns which varied in scale, size, and conservation objectives. The program was implemented in 1993 with the goals of determining the historical role and contemporary effects of fire on the islands, and if fire can be effectively used as an ecological restoration tool. This adaptive management framework is essential because of lack of knowledge of fires role in the islands, and also because feral animals and historic human land use practices have drastically altered much of the composition, structure, and functional elements of the islands. Components of the burn program include: 1) two experiments focusing on management of an alien plant species (fennel, *Foeniculum vulgare*) that differed in scale, objectives, and complexity; 2) a small-scale experiment analyzing differences in grassland species composition as a result of different timing and frequency of burns; 3) a series of large-scale burns analyzing patterns of vegetation succession and structure in grasslands following fall burns; 4) a large-scale burn in grasslands focusing on differential patterns of composition and structure of plant and animal groups in grasslands as a function of fire return interval; 5) a large-scale burn implemented to enhance seedling regeneration in Bishop pine (*Pinus muricata*) forests; 6) laboratory experiments testing the effect of different temperatures on Bishop pine cones opening during a fire, and, 7) a fire history study.

In the small-scale fennel burns, burning coupled with herbicide (triclopyr-Garlon 3A) spraying reduced fennel cover 90%-100%. Burning and spraying was the only treatment where an increase in native species occurred (the other treatments were burn only and spray only). This information was crucial to development and implementation of a larger scale management program focused on multi-trophic level function. In the large-scale grassland burns, total species richness and diversity tended not to change systematically as a result of fall burning, but the postburn composition of the burned areas was significantly different from that of preburned conditions and unburned controls. Burned and unburned plots were dominated by alien grasses in all years, while the guild that tended to show the greatest positive response to the burns were annual forbs. The richness and abundance of native species tended to either increase or remain unchanged as a result of the burning. Small-scale experimental manipulation of the season and frequency grasslands were burned indicate that a greater positive response of native species occurs in spring burns. This information was used to develop a large-scale spring burn examining multi-trophic level effects with fire-return interval as the main treatment. Burns in the

Bishop pine stands led to an increase in pine seedling production, although seedling survival between burned and unburned sites did not differ significantly. The laboratory studies indicated that the number of scales opening and releasing seeds on a cone would increase up to a temperature of 130^0 C, then reach a threshold. The fire history study indicates fire was a relatively frequent occurrence on Santa Cruz Island, although no evaluation can be made in regards to the extent, seasonality, intensity, and other components of the fire regime.

The dynamic interplay between small-scale experiments and large-scale management programs has been the crucial element in the development of the fire program in Channel Islands National Park. Using small-scale experiments to aid in the design of larger scale management burns has allowed us to systematically increase the complexity of the ecological questions we ask and the management goals we hope to achieve. The first five years of the program were designed primarily around a pattern analysis of fire effects. Over the next 10 years, the focus will be on identifying the processes leading to the patterns we observe, and understanding the mechanisms driving the processes.

Keywords: alien species, diversity, fennel, grasslands, fire, prescribed burns, restoration, Santa Cruz Island

Introduction

The role of fire in parks, preserves, and other wildlands has been a topic of great interest to biologists and resource managers. In the early 1900's fire was generally seen as a destructive force that threatened human lives, property and natural resources. Consequently, suppression became the primary fire management practice for federal, state, and local agencies. However, fire suppression led to undesirable and unintended effects on natural communities, including buildup of fuels, altered ecosystem properties, and changes in species composition (Lee 1977, Kilgore 1976, Flannigan and Van Wagner 1991). Consequently, the interaction of these effects led to very large, intense wildfires that were more frequent, difficult and expensive to suppress than in the past. This led to the recognition that fire was a natural process in many ecosystems, and beginning in the mid-1960's a re-evaluation of fire management practices led to the inclusion of natural fires in many wilderness areas (Mutch 1995, Biswell 1999).

Prescribed natural fires were originally seen as being the most desirable alternative for reintroducing fire into wild areas (Mutch 1995). In many systems though, prescribed natural fires were not always appropriate or, in and of themselves, could not meet management goals (Husari 1995, Mutch 1995). As a result, prescribed burning was used to augment or replace natural fires. The original focus of both prescribed burning and prescribed natural fire programs was to reduce fuel loads and manipulate species composition. A large proportion of research was focused on reconstructing historic fire regimes because it was commonly believed that closely mimicking natural regimes would be the most effective way to create desired ecological conditions. However, as Whelan (1994) points out, achieving conservation goals may require that fire not be used in the context of a natural regime.

Prior to the late 1980's, prescribed fire was typically used as a management tool in relatively large, intact ecosystems that had not been severely fragmented or reduced in total area (Lopoukhine 1993, van Wagtendonk 1995). These areas were usually forests or shrublands, and were dominated by native species. Examples include large parks, forests, and wilderness areas such as Yellowstone National Park, Yosemite National Park, Sequoia Kings-Canyon National

Park, and the Selway-Bitteroot National Forest (Brown et al.1995, Kilgore and Nichols 1995). But as the field of restoration science developed, fire began to be used less as a way to reduce fuel loads than a way to potentially recreate the natural structure, composition, and function of badly degraded systems. Many of these restoration areas do not fit the more traditional types of wild lands where much of fire research and management had been focused, and are often dominated by non-native species (Parsons and Stohlgren 1989, Dyer et al. 1996, Kwilosz and Knutson 1999, Klinger and Messer *in Press*). These included remnant prairies in the Great Plains, grasslands in California, and riparian areas in the arid southwest.

The use of fire in a restoration context has increased the complexity of how prescribed fire programs are conceived, designed, and evaluated. The philosophy of most agencies is that prescribed fire is restoration of a natural process. But beyond the general goal of process restoration, specific evaluations must be made of just how fire is effecting different components of the target system. There may be concerns over effects on rare and endangered species (Greenlee 1997), ecosystem processes (D'Antonio and Vitousek 1992), and non-native species (van Wilgen et al. 1990). In restoration programs it is not enough to evaluate whether succession patterns are desirable or not, but whether fire is producing appropriate patterns in abundance of native species (D'Antonio and Vitousek 1992). The more degraded an ecological system is, the more difficult it is to achieve these goals.

Numerous publications exist on outcomes of individual parts of burn programs, but there are few published case studies of programmatic prescribed burn programs, especially in an adaptive management context (van Wilgen et al. 1990). As the use of prescribed fire increases in scope and the goals of burn programs increase in complexity, it becomes important to make case studies available for land managers to review. It is our goal in this paper to discuss the prescribed burn program that was developed and implemented on Santa Cruz Island. We will explain the rationale for the reintroduction of fire to the island, discuss the general framework of the program, review outcomes from specific studies, and evaluate the strengths and weaknesses of the program. We will conclude with an assessment of which parts of the program can probably be generalized to parks and preserves on the mainland.

Ecosystem Features and Land Use History of Santa Cruz Island

The ownership of Santa Cruz Island has changed a number of times since Europeans first settled it in the early part of the 19th century. The Nature Conservancy (TNC), a private non-profit international conservation organization, presently owns the western 90% of Santa Cruz. TNC acquired an interest in the island in 1977 and assumed full ownership in 1988. They are the first owners to manage the island primarily as a natural reserve. The eastern 10% of the island is part of Channel Islands National Park (CINP), who owns or manages the four other northern Channel Islands (Santa Rosa, San Miguel, Santa Barbara, and Anacapa). Consequently, TNC holdings are considered an in-holding within CINP. Organizational Memorandums of Understanding have been developed that outline the similar management goals of the two organizations and allow formal cooperation on resource management programs.

Santa Cruz is the largest of the eight California Islands. Approximately 250 km^2 in area, it is located 32 km off the coast from Santa Barbara. The main axis of the island is east-west trending, and is characterized by two mountainous ranges and a central valley that parallel the main axis. The highest point on the island is above 700m, and the topography is extremely rugged. The geology is varied, with different formations scattered among one another rather than in large, homogenous blocks.

The climate is Mediterranean, with a mean annual summer high temperature of 28^0 C, and mean annual winter low of 4^0 C. The shape of the island and the maritime environment are responsible for creating two different temperature regimes. The interior valley of the island tends to experience greater daily ranges in maximum and minimum temperature than the coastal sections. Rainfall patterns are highly seasonal, with 80% occurring between November and April. The 90-year mean average rainfall is 50 cm.

The primary plant communities on the island are grasslands, chaparral, woodlands, coastal scrub, and pine forest. Grasslands comprise over 50% of the island's area and are dominated by alien grasses (*Lolium multiflorum, Avena* spp., *Bromus* spp., *Vulpia myuros*), with an interspersion of native grasses (*Nasella pulchra*, *N. lepida*, *Hordeum brachyantherum*). Chaparral occurs on about 30% of the island and is dominated by scrub oak (*Quercus pacifica*), toyon (*Heteromeles arbutifolia*), and manzanita (*Arctostaphylos* spp.). The dominant sub-shrubs in the understory are monkey flower (*Mimulus longiflorus* and *M. flemingii*) and *Baccharis plummerae*. Woodlands are comprised of an overstory of scrub oak and coastal live oak (*Q. agrifolia*), with relatively small amounts of sub-shrubs and an understory of grasses and forbs. Coastal scrub is dominated by native shrubs (*Artemisia californica, Baccharis pilularis, Hazardia squarrosus, Rhus integrifolia*), alien grasses, and a combination of native and alien forbs (*Atriplex semibaccata, Dichelostemma capitatum, Erodium spp., Sanicula arguta, Sisyrinchium bellum)*. Pine forests occur in three disjunct areas of the island, and vary in structure and species composition (Ostoja and Klinger *In press*). In general, they are chaparral areas with an overstory of Bishop pines (*Pinus muricata*).

Santa Cruz has a relatively high percentage of endemic plant and animal species. There are 42 plant species endemic to the Channel Islands, eight of which are confined strictly to Santa Cruz (Junak et al. 1995). These species exhibit a wide range of distribution and abundance patterns, and are found across the island in all of the plant communities. Thirteen species are listed by the state or federal governments as either threatened or endangered. Two of these species are presumed to already be extirpated (Junak et al. 1995).

Santa Cruz Island has a long history of human habitation, dating back approximately 7,000 years (Glassow 1980). A great deal is known about the cultural aspects of the Chumash who inhabited the island up to the early 1800's, but relatively little is known of their impacts to the system (Cushing 1993). A great deal more is known about the effects that ranching and agricultural practices had on the island over the last 150 years. Livestock (sheep, cattle, and pigs) were ranched on the island since at least the 1830's, and there were extensive vineyards in the Central Valley by the latter part of the 1880's. Most of the changes in landscape features and biological communities were related to these land use patterns.

The Channel Islands have been severely impacted by feral animals, primarily sheep (*Ovis aries*), goats (*Capra hircus*), and pigs (*Sus scrofa*) (Coblentz 1977, 1978, 1980, Van Vuren 1981, 1984). These impacts were especially pronounced on Santa Cruz Island, where more than 50,000 sheep were estimated to be on the island in the 1890's. There were estimated to be 20,000 sheep on the island in the 1980's, which was more than double the maximum stocking rates of mainland sheep operations (Van Vuren 1981). Over one-third of the island was classified as being heavily impacted (Van Vuren 1981). This resulted in an increase in bare ground and subsequently higher erosion rates, decreased herbaceous vegetation, reduction and modification of shrub communities, and a decrease in abundance and diversity of birds (Brumbaugh 1980, Hobbs 1980, Hochberg et al. 1980, Minnich 1980, Van Vuren 1981). During the 1980's The Nature Conservancy (TNC) undertook a program to eradicate feral sheep from the 90% of Santa Cruz which it owned. This program was successful, and by the end of the decade the only sheep on the island were on the eastern 10% (Schuyler 1993). Following the sheep eradication, the 7,000 head of cattle that had been ranched on the island were rounded up and shipped off.

In addition to feral animals, alien (non-native) plants have also had a significant impact on the islands biotic communities. Aliens comprise a significant proportion of the vascular plant species on California's Channel Islands, but their influence extends beyond the number of species. The cover of some plant communities on the Channel Islands is dominated by alien plants (grasslands, coastal scrub) (Halvorson 1994), and there is evidence that proportionally more alien species are now more widely distributed and have greater local densities than native species (Klinger 1998). On Santa Cruz, approximately 25% of the flora is non-native, but over 75% of the herbaceous cover in grasslands and woodlands is comprised of alien grasses. Fennel (*Foeniculum vulgare*), a perennial species from the Mediterranean, has altered the structure and composition of grasslands across 10% of the island (Beatty 1991, Brenton and Klinger 1994, Klinger 1998).

Fire Program Development and Design

The development of the fire program on Santa Cruz began in 1990. Representatives from TNC, CINP, the California Department of Parks and Recreation, and the University of California held a meeting to assess the rationale and feasibility of implementing a prescribed fire program on the island. The goals of the meeting were to evaluate the historic role of fire on the island, develop a suppression plan that was compatible with the conservation and management goals of TNC and CINP, and determine whether a fire program should primarily consist of prescribed burning or prescribed natural fire.

There was no detailed data regarding the historic role of fire on the Channel Islands, but charcoal deposits and fire scars indicated that it had occurred on the islands, and in all likelihood had been a significant natural process on Santa Cruz. Little was known about aboriginal burning on the islands, but historical and contemporary records indicated that large fires had been uncommon over the last 150 years. This was due to suppression practices by the ranchers, and also because of the lack of fuels due to the severe overgrazing which had occurred on the island. In addition, the only study related to fire on the islands indicated that plant species had similar fire adaptations as those on the mainland (Carroll et al. 1993). Finally, seedling survival and recruitment of Bishop pines was extremely low in two of the three stands of pines on the island. Adult mortality was also very high in these stands, primarily because of infestation by pine bark beetles (*Ips* sp.). Since Bishop pines are a strongly serotinous species and beetle infestations are low in stands of closed-cone pines which burn regularly (Holland and Kiel 1995), the conclusion was that the low recruitment and high mortality patterns were likely related to the lack of fire in the stands.

It was the consensus of the participants that there was both biological and organizational justification for a fire program on Santa Cruz. The management goals of both TNC and the National Park Service include the reintroduction of natural processes on lands that they manage, and the evidence that did exist indicated fire had occurred historically on Santa Cruz Island. There was indication that fire suppression had impacted at least one of the natural communities on the island. It was recognized that fire on the islands would become more frequent because of the increase in the extent and density of fuels, accidental ignition, and reduced suppression efforts (Carroll et al. 1993, Wells 1991). However, because there was so little data on the historic fire regime and effects on the island communities were unknown, the program should consist of prescribed burns rather than prescribed natural fire. A suppression plan was produced that emphasized "light on the land" tactics, and established the context of a prescribed burn program for the island (Wells 1991).

Despite a clear rationale for the appropriateness of fire on Santa Cruz, there were a number of management issues that needed to be considered in the development of the program. These included:

1. The high proportion of non-native plants and animals on the island.
2. The high proportion of endemic plants and animals on the island.
3. The residual effects from the historic land uses on fire.
4. Succession patterns following the removal of feral sheep and cattle from the island.

The greatest concerns were whether fire would be beneficial to native species and endemics, and would the relative abundance and impacts of alien species increase in burned areas (especially areas where aliens were relatively uncommon, such as chaparral and pine forests). In essence, fire could not be looked at simply as an ecological process to potentially be restored to the island ecosystem; it could either be a restoration tool, or a disturbance factor with unpredictable effects on a heavily altered system.

Although prescribed burning had been suggested as a tool for restoring native species to grassland communities and had been tested in several areas of California (Parsons and Stohlgren 1989; Dyer et al. 1996), the results of these studies were not clear-cut. In the foothills of the Sierra Nevada the biomass of alien annual grass was reduced and the biomass of both alien and native forbs increased following three successive burns (fall or spring), but these effects were transient and were not sustained beyond the burning treatments (Parsons and Stohlgren 1989). At the Jepson Prairie, Dyer et al. (1996) found that recruitment of *Nasella pulchra* was relatively high in burned areas but was also highly dependent on variations in annual climatic conditions. On the Santa Rosa Plateau in the Coast Range of southern California, Wills (Unpubl. data) found greater levels of *Nasella pulchra* in burned areas.

The role of fire on Santa Cruz had two aspects then: the first as a naturally occurring process under ecological conditions that varied drastically from historic ones, and the second as a restoration tool. These aspects were not necessarily exclusive, but had to be evaluated by different criteria. This required the development of a program within an adaptive management framework (Holling 1978); the ecological effect of single-fire events would be studied at meaningful spatial scales, while small-scale experiments would be used to develop restoration protocols for application at larger spatial scales. An understanding of single-fire events would enable management of unplanned fires within acceptable ranges of spatial and temporal variability. Additionally, outcomes from restoration projects that used fire could be tested against explicitly stated goals. If restoration projects did not meet these goals, then a different series of experiments could be developed to test why they did not meet expectations.

From this, two inter-related sets of goals were defined for the first phase of the fire program. The ecological goals were to: 1) Develop an understanding of the historic fire regime on the island; and 2)Determine the effect of single-fire events on the structure and composition of the island's communities.

The restoration goals were to:
1. Determine whether the distribution and abundance of native species increased relative to alien species as a result of burning;
2. What the effect of fire was on the demography of selected native species;
3. Whether populations of selected alien species could be controlled by using fire.

Based on these goals, four objectives were set for the next 5-10 years. These included:
1. Conducting a series of moderate sized burns (100-500 ha) mimicking single fire events in grasslands and pine forests;
2. Testing fire as a potential way of controlling fennel;
3. Determining the effect of varying fire regimes on grassland communities and bishop pine;
4. Initiating a study of the islands fire history for the last 1,000-20,000 YBP.

We chose to conduct the studies in grasslands for several reasons:
1. Their extensive distribution on the island;
2. The alien species have had the greatest relative impact in this community;
3. The light, flashy fuels make them the most likely areas for a fire to either start or be carried across an extensive part of the island;
4. TNC was studying the effect of fire in grasslands on other preserves in the state.

Pine communities were selected because of the high adult mortality and low recruitment of Bishop pines. Fennel was targeted because of the ecological impacts it had on other communities and the management interest in controlling its populations (Beatty and Licari 1992, Brenton and Klinger 1994, Dash and Gliessman 1994). It was decided that fire should not be tested in chaparral, coast scrub, and woodland communities because of the high regeneration of woody species following the removal of cattle and feral sheep.

Program Implementation and Outcomes

The first phase of the fire program was implemented in the fall of 1993 with a small-scale fennel burn and a moderate sized burn in grasslands on the southwest side of the island (Table 1). A total of seven projects and 151 individual burns were conducted between 1993 and 1999. Burns were done over three spatial scales; small (<1 ha), medium (> 1 and < 500 ha), and large (> 500 ha). Over 95% of the burns were small scale (n=144), and were done as part of an experiment testing the effect of fire frequency and burn season in grasslands. Small scale burns totaled 0.36 ha in area. Three moderate sized burns done in grasslands between 1993-95 totaled 1,200 ha, with the largest of these approximately 490 ha (see below). The total area burnt in all of the projects was approximately 3,250 ha, with slightly over 2,400 ha burnt twice in grasslands on the southwest side of the island. The shortest fire return interval on any area of ground > 1 ha was four years (fennel control project and fire return interval in grasslands).

In terms of proportion of total area of a community type on the island that was burned in the projects, grassland comprised 6.2%, pine forest 25.6% and fennel 10%. The total area of the island that was burned was 7.9%. A fire ecology and burn-leader training workshop sponsored by TNC and CINP was held in the fall of 1997. An additional 12 ha of grassland and 2 ha of coastal scrub was burnt as part of the training exercise, but data were not collected in these sites. Unpredicted Santa Ana winds led to an additional 100 ha being burned in the east stand of pines in 1994, but this area was scheduled to be burned in 1995 and monitoring plots had already been

Table 1. Studies of fire effects on Santa Cruz Island, 1993-1999. *Community* is the community type burned, *Burn Years* are the years burn were done, *Size* is the total area burned in the project, *Study Duration* is the period of time that fire effects were monitored, and *Level of Study* is the biological level of organization the study focused on (population of single species, community, and ecosystem)

Project	Community	Burn Years	Size (ha)	Study Duration	Level of Study
Single Fire Event	Grasslands	1993, 1994, 1995	1200	1993-1998	Community
Fennel Control	Fennel	1993	2	1993-1997	Community & Population
Single Fire Event	Pine Forest	1994	200	1994-Present	Community & Population
Fire Frequency and Season	Grassland	1996, 1997, 1998	<0.01	1996-2001	Community
Bishop Pine Serotiny	Laboratory	No Burns	0	1996-1998	Population
Fennel Control	Fennel & Grassland	1997	250	1996-2001	Population & Ecosystem
Fire Return Interval	Grassland	1999	1600	1997-2004	Ecosystem

established and sampled. No other significant escapes occurred on any of the other burns. Two wildfires occurred on the island between 1990-99. One consumed approximately 100 ha at Chinese Harbor in the summer of 1990 (prior to the development of the burn program on the island), and the other burnt <2 ha at the University of California Field Station in the spring of 1991. Neither burn caused any significant structural or ecological damage.

The study design, objectives, and preliminary results for each of the individual projects is outlined below.

Single Fire Events in Grasslands

Three prescribed burns of 270, 340, and 490 hectares were done in grasslands on the southwest side of the island between 1993 and 1995. The main goals of the burns were to begin an evaluation of the effect of fire as an ecological process in the island's communities, and to determine if fire would enhance native plant species distribution and abundance.

The effect of fire on herbaceous and woody species was monitored by sampling in the spring preceding each burn and then each spring for three years after. Data were collected in 10 burned and 10 control grassland plots (matched by year) for each of the three treatment areas. The data included estimates of species richness, cover of herbaceous and woody species, and density of woody species.

Total species richness and diversity tended not to change systematically as a result of burning, but the postburn composition of the burned areas was significantly different from that of preburned conditions and unburned controls. The response to burning varied between different vegetation guilds and within the different burn areas. Burned and unburned plots were dominated by alien grasses in all years, while the guild that tended to show the greatest positive response to the burns were annual forbs. By the third year postburn, the burned plots were beginning to return to a composition similar to preburn conditions. The richness and abundance of native species tended to either increase or remain unchanged as a result of the burning. Shrub cover in burned areas decreased, but density of adult and seedling shrubs did not change significantly.

A detailed description and analysis of this study is given in Klinger and Messer (*in press*).

Fire Return Interval in Grasslands

Fire regimes are characterized by seasonality, frequency, return interval, intensity, duration, spatial extent and spatial continuity (Whelan 1994). To understand how different components of the fire regime affected grassland communities on the island, a study of return interval was initiated in 1996. This project used the three areas in the southwest grasslands that had been burned from 1993-95, with one additional 200 ha adjacent to the other three that was burned in 1996. All four areas plus an additional 200 ha were burned in the late spring of 1999, giving return intervals from 3-6 years at one-year increments.

Because each return interval was unreplicated, a BACI design (Osenberg et al. 1994) was used for the study design. Data collection began three years prior to the burn in 10 plots in each burned area and an unburned control area. Besides composition and structure of vegetation, data were also collected on soil characteristics and the distribution and abundance of small mammals, birds (breeding and wintering populations), and reptiles. Logistical problems required that the reptile sampling cease in 1999, but all other aspects of the study are underway.

Fire Frequency and Burn Season in Grasslands

This experiment was implemented in 1996 to determine the effect that different combinations of season of burn and return interval had on composition and structure of grasslands. Data from two other TNC preserves in California (Santa Rosa Plateau and Jepson

Prairie) indicated that native grasses responded better to burns done in the late spring (prior to seed dehiscence) than in the fall. Data from the grassland burns done on the island from 1993-95 suggested that single-fire events were responsible only for short-term changes in species composition, and topographic features had the greatest long-term influence on species composition. If fire was to be used as a restoration process, then manipulation of fire regime would be necessary (Klinger and Messer, *in press*). Because of unknown and potentially undesirable effects (disruption of shrub succession, creating conditions favoring proliferation of alien species) resulting from manipulating the fire regime, the experiments were done on a small scale.

Four blocks consisting of twenty-four 25-m^2 plots were established in a mid-elevation grassland in the central part of the island. Each block was arranged on a different aspect (north, south, east, and flat). There were eight treatment conditions, with three replicates/treatment in each block. The treatments were plots burned once, twice, and three times either in the late spring or mid-fall, and unburned spring and fall control plots. The burns were done from 1996-98, with a fire return interval of one year for the plots with multiple burns. Data on species composition, cover, and height was collected prior to burning, with post-burn data scheduled to be collected annually until 2001.

A preliminary analysis of data from 1996-99 indicates that species richness is lower and species evenness is greater in plots burned three times during the spring than in unburned plots. The decrease in species richness is primarily due to the disappearance of less abundant alien grass species. There is no significant change in diversity or composition between plots burned during the fall and unburned controls. We emphasize that these results are preliminary, and should be recognized as such and interpreted very carefully. Detailed analyses will be conducted after 2001.

Single Fire Events in Bishop Pine Forests

A prescribed burn was done in the east stand of pines to enhance seedling regeneration and determine the effect of a single fire on community structure and composition. Bishop pine cones are sealed by a resinous coating. This coating melts when exposed to sufficient heat, resulting in the release of mature seeds. Because of this reproductive strategy, and because these species occur in maritime and insular regions dominated by chaparral and scrub communities, it is assumed that they are adapted to historical periodic fires and may even require fire for successful regeneration (Vogl 1973; Zedler 1986).

The east stand was selected because regeneration of shrub and tree seedlings in the north stand was already high, and fire would have probably led to mortality of the seedlings. The west stand of pines had low seedling regeneration and very high mortality of adult Bishop pines, but the extremely high fuel load made control of a fire in the stand problematic. Control of a prescribed burn was considered to be less difficult in the east stand. Seedling regeneration in the stand was low and mortality of adult trees very high; only five adult trees and nine seedlings were found in a reconnaissance done in the main part of the stand in the fall of 1993. Two of the five adult trees showed signs of stress from the bark beetle.

Three 0.25 ha (50m x 50m) macroplots were established in each of three conditions; control (unburned), 1994 burn, and 1995 burn. The control plots were in a disjunct part of the stand (Los Pinos del Sur) three km away from the main section (Chinese Harbor). Adult mortality of Bishop pines was lower in the control area. However, seedling regeneration was low and the species composition and stand structure were similar to the main part. We recognized that the sampling design was pseudoreplicated, but it would have been extremely difficult to keep fire out of the relatively small control macroplots if we had spatially replicated them within the burn area. We chose not to use a BACI sampling protocol for several years prior to burning because of the alarming decline in adult trees.

Three randomly located 30m x 2m transects were established within each macroplot. There were five randomly located $1m^2$ quadrats along each transect. A complete species list was made for each macroplot, and the density, height, and cover of all woody species was recorded along each transect. The cover and height of all herbaceous species were recorded in each $1m^2$ quadrat. Sampling was done in all macroplots prior to burning in 1994, and then annually each spring or summer through 1998. Sampling is scheduled to occur at three-year intervals through 2012.

Because the initial density of seedlings was low in each macroplot, all seedlings found in the stand from 1994-96 were marked and monitored for growth and survival. Monitoring was done twice each year from 1995-98. Seedling density was estimated using variable-area transects (Parker 1979).

The burn in the 1994 area (80 ha) was completed successfully in early November under cool and moist conditions (15^0 C, 72% RH, northwest winds 5-10 mph). However, unpredicted Santa Ana winds occurred overnight and the next morning a spot fire started about 0.4 km inside the 1995 burn area (120 ha). The weather conditions were warm and dry (29^0 C, 12% RH, northwest winds 20-50 mph), and the entire 1995 area burned within a matter of hours. This resulted in three macroplots each in three conditions; an unburned control area, a prescribed burn area with fire of low intensity, and a wildfire area of high intensity.

That winter a strong El Nino event occurred, with the third greatest amount of rainfall falling on the island in the 90 years since records had been kept (117 cm vs. 50-cm 90-year average). The rainfall appeared to have a pronounced effect on seedling regeneration. In 1995, the density of seedlings was low, and they occurred almost exclusively in patches in flat areas, but not on slopes. This indicated that there had been significant loss of seeds due to the heavy rains. There was a significant increase in seedling density in 1996, indicating that a second burst of germination had occurred. There was no significant difference in seedling density between the burned and unburned areas in 1995, but seedling density was significantly greater in the wildfire area than in the prescribed burn area or unburned area from 1996-98. Seedling survival was high in all three areas (> 90%).

A preliminary analysis done in 1997 indicated that species diversity (richness and evenness) increased significantly in the burned areas. This was primarily due to the number of fire-following herbaceous species, mainly forbs. One endemic forb, *Heliathemum greenei*, dominated the herb layer in the wildfire area for the first three years after the fire. The regeneration of all shrubs was high, especially *Quercus parvula* and *Lotus scoparius*.

As with the experiments done on season and frequency of fires in grasslands these results are preliminary, and should be recognized as such and interpreted carefully. Because the study was pseudoreplicated, inference of these results to the two other stands on the islands would be inappropriate. They should only be used as a general framework for generating hypotheses if prescribed burns are conducted in the future in the north or west stands.

Serotiny of Bishop Pines

It is assumed that the level of serotiny in closed cone pines should mainly depend on the fire regime (Holland and Keil 1995), but other factors such as local environmental conditions or morphological traits may influence serotiny as well (Perry and Lotan 1979, Borchert 1985). In the long-term absence of fire, a stand of trees may become senescent as individual trees lose vigor and produce fewer and fewer new cones. Observations we made in the stands over the last eight years indicated that cones were opening and seedling regeneration was occurring in all three despite the absence of fire. How this compared to potential regeneration following a fire was unknown. In addition, the form of the response of the cones opening at different temperatures was unknown. Several possibilities existed, including a direct linear response with increasing temperature; a linear response up to a threshold temperature (beyond which relatively few scales opened); or a stepwise response to multiple ranges of temperatures, where the percentage of scales that opened remained constant within each range.

The goal of this study was to identify the response of Bishop pine cones to varying temperatures, determine whether this pattern was characteristic of the different stands, and relate morphological characteristics of the cones to the patterns we observed in the scales opening (Ostoja and Klinger, *in press*). We hypothesized that if morphology or local environmental conditions had modified the response of the cones to the historic fire regime, then we would expect to see different patterns of scales opening between the different stands or among cones of different sizes. Alternatively, if adaptations to the historic fire regime were more important than adaptations to local conditions, then we did not expect to see differences between stands in the proportion of open scales at different temperatures.

Fifty cones from each stand were tested at five different temperatures (30^0, 80^0, 130^0, 180^0, 230^0 C). The number of open scales/cone was related positively to temperature, but there was no significant difference in the percentage of scales opening at temperatures greater than 130^0. Morphological differences in the cones existed between the stands, but there was no significant difference between the stands in the percentage of scales opening at the different temperatures. At intermediate temperatures (130^0 and 180^0), the percentage of open scales/cone was positively related to cone size, but negatively related to the number of scales/cone (Ostoja and Klinger, *in press*).

The results indicate that local adaptations are not directly influencing serotiny patterns in the island's Bishop pines, and that variations in fire behavior can lead to potentially different patterns of postfire regeneration. It has been suggested that without fires, closed cone conifers may be at risk of succumbing to parasites and be replaced by other species (Holland and Keil 1995). Because Bishop pines will reproduce without fire, this scenario would probably only happen under extreme conditions. However, it is clear that reproduction in Bishop pines is greater at higher temperatures, and since the community they occur in is characterized by species with fire adaptations (Carroll et al. 1993), it would appear as if some degree of burning within the stands would not have any severe detrimental effect on the species or the community (Ostoja and Klinger, *in press*).

Fennel Control

Phases I and II

Fennel is a perennial herb introduced to Santa Cruz Island from Europe in the late 1800's. It now dominates a substantial proportion of grasslands throughout central and northeastern Santa Cruz Island and is continuing to expand its range (Beatty and Licari 1992, Klinger 1998). Though it has been on the island for over 100 years, the current distribution and abundance did not occur until grazing pressure from cattle was removed in the 1980's and a 5-year drought ended in 1991 (Brenton and Klinger 1994). The ecological effects have included alteration of community structure, shifts in community composition, and displacement of native herbaceous species (Klinger 1998). Because of these effects, management of fennel became a high priority project for TNC.

A state-transition model of fennel expansion and control in grassland/coastal scrub and riparian communities was developed (Brenton and Klinger 1994), and two different programs studying different control methods have been underway since 1991. In a study conducted between 1991-94, Brenton and Klinger (1994) examined the effect of spraying fennel with different formulation/concentration combinations of the herbicide Garlon@, manual cutting, and season of herbicide application. Spraying in the wet season was found to be the most important factor for reducing fennel cover, where reductions of 50-90% were observed (Brenton and Klinger 1994). The areas where fennel was removed were replaced predominantly by alien grasses (Brenton and Klinger 1994). Gliessman and his students have studied the relative effectiveness of manual removal (digging), mowing, and spraying the herbicide Roundup@. Herbicide spraying was found to be the most effective way of controlling fennel, but they also found alien grasses to be the primary ground cover after fennel was removed and that native species were possibly harmed by spraying (Dash and Gliessman 1994).

In contrast to the relatively indiscriminate effects of Roundup@, Garlon@ is a relatively specific broad-leaf herbicide. There was no evidence that Garlon@ had negative impacts on native species at the concentrations used by Brenton and Klinger (1994), so it was suggested that using fire in conjunction with Garlon@ would increase the species richness and abundance of native species. Klinger and Brenton (unpubl. data) tested the null hypothesis that there would be no difference in the relative effectiveness of fire and herbicide (triclopyr) as fennel control methods, and there would be no change in the relative abundance and species richness of native herbaceous species. Using a nested design, four experimental conditions (burned and unsprayed, unburned and sprayed, burned and sprayed, and unburned and unsprayed) were established in eight 0.40 hectare macroplots. Data was collected on the cover and height of fennel, and all other plants occurring with fennel, from the spring of 1993 (pre-treatment) to the spring of 1997. Four of the 0.40 ha macroplots were burned in the fall (November) of 1993. Garlon@ was applied to the burned & sprayed and sprayed treatment macroplots in the spring of 1994 and 1995.

Fennel was virtually eliminated in treatment areas that were burned and sprayed, and there was a significant increase in the mean cover and number of native herbaceous species. Fennel cover was drastically reduced in areas that were sprayed and unburned, but there was no significant change in the cover or number of native species. Alien herbaceous species dominated the cover and species richness in both burned and sprayed and the unburned and sprayed areas after fennel cover was reduced. Relative to the unburned and unsprayed areas, there was virtually no change in the species composition or structure that were burned and unsprayed. Fennel cover and height increased and native species richness and cover decreased in the control plots.

Phase III

Although the results of the burning and spraying experiments were encouraging, there was no data on the effect of fennel removal on functional aspects of the ecosystem, or on structure and composition of animal assemblages. A six-year study was initiated in 1996 looking at the effects of removal of a dominant exotic plant from an ecosystem. The primary goals of the project were to; 1) identify the changes in diversity, structure, and composition of plant and animal communities following the removal of fennel, and, 2) following the removal of fennel, identify the interactions between changes in habitat structure, productivity, and trophic relations that determine the patterns that were observed in the diversity, structure and composition of plant and animal communities.

A nested design was used to monitor changes in the structure, function, and composition of 4 groups of organisms in two control and three treatment conditions one year before and four years after removal of fennel. The groups of organisms are vascular plants, selected insect species, salamanders, and lizards. Data for these groups is being collected from an equal number of randomly located 0.1-hectare macroplots in each of five conditions (Table 2). Burning was done in 1997 and spraying completed in the spring (May) of 1998 and 1999. Preliminary results for the program are not available at this time.

Table 2. Experimental design for fennel removal experiment, Santa Cruz Island, California."Macroplots" is the projected number of macroplots in each treatment condition; the number of which may change as a result of a pilot study being conducted during the summer of 1996.

Habitat/Treatment Condition	Designation	Macroplots
Fennel-Burned & Sprayed	Treatment	5
Fennel-Burned, Sprayed and Seeded	Treatment	5
Grassland-Burned	Treatment	5
Grassland-Untreated	Control	5
Fennel-Untreated	Control	5

Fire History

A fire history study was initiated in late 1997. A preliminary analysis indicates that fire predated humans and was an important process on the island (Anderson, pers comm). Fire return interval appears to be similar to the mainland, and several notable periods of elevated fire have been documented. Anderson (pers comm) emphasizes that these results are not final, and additional samples have yet to be analyzed.

Program Evaluation

The initial results of the prescribed fire program on Santa Cruz Island are encouraging, but the outcomes from both an ecological and a conservation perspective are inconsistent and unpredictable. The program demonstrates how spatial and temporal variability will have direct and indirect effects on ecological processes, and how this will effect programmatic outcomes.

Although single fire events in grasslands led to an increase in richness and abundance of native herbaceous species, these communities continued to be dominated by alien grass. Preliminary evidence indicates that multiple late spring burns may lead to an increase in species diversity of natives, but the effect of different fire prescriptions (which represent variation in fire regimes) on species composition remains unresolved. Variation in weather between fires, topography, historical land use, initial composition and structure of vegetation in the burn areas, and variation in fire behavior (e.g. intensity and rate of spread) all potentially influence response of vegetation to burns.

Determination of the effect of fire interval on succession patterns is probably the most important component of the fire regime that could be studied over the next 10-15 years. Shrub regeneration on Santa Cruz Island since the removal of feral sheep and cattle has been high, and a management goal is to reduce the extent of grasslands on the island by 5-10% over the next 15-20 years (Klinger et al. 1997). Burning too frequently in grasslands may not allow shrubs to move into grassland areas, but suppressing fire will probably prevent them from establishing extensive stands as well (Schultz et al. 1955). Studies should focus on the optimal combination of season, frequency, and return interval to not inhibit native shrub establishment, and to allow native herbaceous species to increase in abundance.

Allowing fires to occur in the Bishop pine stands should be approached with caution. As we discussed above, regeneration of woody species in the north stand of pines is high and a fire at this time would probably have a detrimental impact. Although we did not observe an increase in alien species invading the burned parts of the east stand, the limitations of our study in that stand make inferences of this pattern tenuous. From an ecological perspective, long-term fire suppression is clearly an undesirable alternative. However, the timing and scale of burns should be carefully considered. Although there are clear ecological reasons for allowing fire in the pine stands, conservation considerations may limit its use to small scale, experimental burns for a number of years (Ostoja and Klinger, *in press*).

The results of the fennel control experiments indicate that fire alone will not reduce fennel cover, but it is an effective way to enhance the action of triclopyr and create conditions more favorable to native species than simply spraying herbicide. Spraying Garlon$^{@}$ without burning first led to a dramatic reduction of fennel, but there was no change in the richness or abundance of native species. Therefore, the most important reason for using prescribed burns in conjunction with Garlon$^{@}$ application is the beneficial effect it has on native species. Even then, these benefits will be incremental. While plots that were burned and sprayed with Garlon$^{@}$ were the only ones where the species richness and cover of natives increased, they were still dominated by alien grasses.

Our results show that the removal of an exotic plant species alone will not necessarily lead to communities dominated by native species. In addition, these experiments have been conducted in grasslands, and using this protocol in coastal scrub communities that have been invaded by fennel needs to be carefully studied (Brenton and Klinger 1994).

The emphasis of the fire studies on the island has been pattern analysis, and it wasn't until 1997 that projects started to be designed to simultaneously look at processes, mechanisms and patterns (Table 3A). Pattern analysis is important as an approach for doing exploratory experiments, and also for generating hypotheses and evaluating outcomes for conservation programs. But actual understanding comes from determining the processes producing the patterns (e.g. relative importance of season of burn, frequency, and return interval), and the mechanisms underlying the direction and rate of the patterns (e.g. change in demographic parameters, seed bank release). When processes and mechanisms are understood, burning protocols can be designed to manipulate these and increase the effectiveness of management programs.

Most of the studies in the first phase of the program were focused on changes in the structure and composition at the community level (Table 3B and 3C). Incorporation of functional aspects into the studies were generally lacking until the second phase of the fire program. Fire may have a significant effect on food web structure (connectance, species diversity, guild structure, guild diversity), so attention will be focused on studies of changes in multi-trophic level relationships in future fire studies on Santa Cruz.

A great deal of importance was placed on achieving positive conservation outcomes from the fire program on Santa Cruz, and the overall goals of conservation management by CINP and TNC center on maintenance or enhancement of natural biological diversity (Klinger et al. 1997). Because ecosystem function can have a great deal of effect on biological diversity, (and vice-versa; Loucks 1970, Tilman 1999), hypotheses regarding the effect of fire on functional aspects of the ecosystem are being incorporated into burns in the second phase of the program (Table 3B and 3C). These include the relationship between productivity and changes in soil texture and chemistry, and whether nutrient levels and productivity are related to changes in soil seed bank. These in turn would effect post-fire composition and structure of communities.

Preliminary results of the fire history study show a long-term pattern of short fire return interval, but it does not reveal other aspects of the fire regime such as season, extent, and intensity. It is likely that the Chumash Indians who inhabited the island burned, but it is unknown how frequently they did this (Timbrook et al. 1982). There have been significant climatic changes on the island over the last 12,000 years, and the landscape changes on Santa Cruz over the last 150 years are well known. For these reasons, it is probably inappropriate to think in terms of the fire program as process restoration. It is likely that the fire regime varied considerably over time, so coupled with the burning by the Chumash and drastic changes in vegetation community distribution, structure, and composition, just what fire as a process should be restored to is problematic. Because of the management goals of CINP and TNC, conservation outcomes are high priority. However, achieving conservation goals may be at odds with a "historic fire regime" (Whelan 1994). It probably makes better sense from an ecological and management sense to try and reproduce the fire effects that give the greatest benefit to native species and communities, rather than try to restore a process to some presumed historical state (Kilgore 1973).

Santa Cruz differs from most mainland preserves in being an island and the severity of the impacts by feral animals. However, the issue of trying to manage or restore a system that has been badly degraded is a general one that goes beyond the island's boundaries. Fire is a process of immense ecological importance, but commitment to a program of fire management will require an understanding that the program will be long-term, phased, and often have complex and contradictory outcomes. Fire suppression has been shown to have many detrimental ecological effects and is a poor practice for conservation management. But expectations of beneficial outcomes from ecological burning programs need to be tempered with the realization that these will rarely be achieved in a cheap and easy fashion. Burning for ecological and conservation purposes will involve a long period of developing a comprehensive understanding of fire as a process in a contemporary rather than historic setting. This will hold for most parks and preserves, whether they are a true island or a habitat island.

Table 3. Programmatic emphasis of the prescribed burn program on Santa Cruz Island, 1993-2000. Phase 1 programs (1993-7) are noted with a (I), Phase 2 (1998-2000) with a (II).

A. Focus of Study

Program	Pattern	Process	Mechanism
Single Fire Events in Grasslands (I)	X		
Season and Frequency of Burns in Grasslands (I)	X	X	
Single Fire Events in Pine Forests (I)	X		
Fennel Control Experiments 1 (I)	X		
Fire Return Interval in Grasslands (II)	X	X	
Serotiny in Bishop Pine Stands (II)	X	X	
Fennel Control Experiments 2 (II)	X	X	X

B. Characteristics Studied

Program	Structure	Composition	Function
Single Fire Events in Grasslands (I)	X	X	
Season and Frequency of Burns in Grasslands (I)	X	X	
Single Fire Events in Pine Forests (I)	X	X	
Fennel Control Experiments 1 (I)	X	X	
Fire Return Interval in Grasslands (II)	X	X	X
Serotiny in Bishop Pine Stands (II)	X		
Fennel Control Experiments 2 (II)	X	X	X

C. Level of Organization Studied

Program	Species	Community	Ecosystem
Single Fire Events in Grasslands (I)		X	
Season and Frequency of Burns in Grasslands (I)		X	
Single Fire Events in Pine Forests (I)	X	X	
Fennel Control Experiments 1 (I)	X	X	
Fire Return Interval in Grasslands (II)		X	X
Serotiny in Bishop Pine Stands (II)	X		
Fennel Control Experiments 2 (II)	X	X	X

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Planning and Use of Prescribed Fire 20 Years, 20,000 Acres, 20 Dollars: A Midway Review of a Successful Prescribed Fire Program on One Private Forest

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Over the past 20 years, the use of fire in the forests of California has become an accepted practice. This has been a cumbersome process which is still ongoing. The reasons for burning and how fire should be reintroduced are still points of contention. When first deciding to use fire as a management tool, I used the simplistic approach. That is, fire has been present for thousands of years, therefore it is an essential component of the Sierra Nevada ecosystem. If we are going to use an ecosystem approach to forest management, fire is required to complete that objective. While the argument to burn is valid, accomplishing this under present day conditions is complicated. A number of factors must first be considered. Since the selected goal is to emulate pre-settlement conditions, the primary question is: what did our forests look like prior to management? This has to be done on a forest type by forest type basis. Then, using science, reintroduce fire into the forest to gain the desired stand conditions. At this point the decision to use fire is straightforward and generally accepted. However, the next steps become not only more difficult, but contentious. Using my fire management program, established in 1980 as an example, I will describe how the use of fire as an ecological requirement has been accomplished.

In 1980 the idea that fire should be considered as an essential component of an ecosystem was not even a consideration. Therefore, that argument was not presented by myself, to either Edison management or to regulatory agencies issuing permits. I needed to modify my justifications to the appropriate agencies in order to proceed with a burn program. My first political ally was wildlife management. Using deer browse benefits as the objective, many thousands of acres were treated with fire in the early 1980's. However, when talking to foresters, I would emphasize tree growth, thinning and natural reproduction as the reasons for burning. More recently, fuels and fuel reduction have become the political buzzwords when citing reasons to burn in our forest. In reality, fire benefits and affects all aspects of an ecosystem, not simply one component such as fuels or wildlife.

The first question always asked is, "How much does it cost?" The costs of this program can be measured in many ways. On private forest lands, expenditures must be tied to either increased income, or other significant benefits such as, increased diversity in wildlife and vegetation, increased tree growth, reduced fire hazard, and increased water flow. In addition, I have never had a budget to burn. The only hard dollar benefits that I can use as a measure is income from future timber harvests. Thus, I must keep the costs below approximately $5 per acre. My other justifications of intrinsic value increases are just that, justifications, but are extremely necessary when dealing with public interests. My question in return is, "How much does it cost to not burn?"

As mentioned earlier, we must first determine what the pre-settlement forest looked like. The evidence is out there, but choosing which observations and research to use is the critical decision that must be made with care, since the variables are so extensive. The physics of fire behavior, on the other hand, is an absolute given. The logical and unbiased evaluation of fire behavior will accurately define how the forest burned under natural conditions. The arguments that indigenous people, cattlemen, and miners used fire, are irrelevant in this discussion since they burned under pre-settlement conditions. We don't know what those conditions were nor the actual timing of the burning. We do know that there is a difference with the fuels. Not only in the loading but the altered composition that has resulted from years of suppression and the introduction of alien species.

As mentioned the mainstay of this scenario is the physical aspects of fire. Fire burned 10,000 years ago just as it does now. That is, fire burns hotter when going up hill, when conditions are dry or windy, and cooler when moving down slope, and in wet, still conditions. Some areas such as grass fields burn several years in a row, while other areas with heavier fuel loads may not have experienced fire for 10 years, and thus burn much hotter. The results of these naturally occurring fires are predictably varied. Results range from open stands of mature trees adjacent to brush fields, to thickets of pine reproduction. The variegated structure of the forests of the central Sierra were sculpted by natural, uncontrolled and very different types of fire. What has changed are the political issues that influence management actions. For example, "old growth" provides emotional barriers to making sound scientific determinations regarding fire and it's relation to pre-settlement forest structures.

To achieve my objective of a pre-management forest structure following 100 years of timber harvesting and fire exclusion required flexibility and opportunistic planning. During the early years of the burn program, most burning was done under a very low profile. This was due to liability, smoke management considerations, and at that time, the public's perception that all fire was destructive. While we are not yet at a midpoint in our program, we do have 20 years and 10,000 acres of burning experience. Some areas have been treated several times with fire. This does not mean that those areas are closer to the desired results than areas treated only once. Some areas need more than five treatments to achieve the desired results. The definition of desired results varies between vegetation types, slopes, exposure, and proximity to residences, and may change over time. Our objectives have already been met, as every resource has benefited from fire. Due to the lack of funds, extensive formal monitoring has not been possible. Roughly speaking, 40% of our measurements are empirical while 60% consists of casual observations. However, as field observations show and photos verify, the results of our burns have been more than satisfactory, and are briefly outlined below.

Fire Protection

In all cases except in extremely cool burns, fuels were reduced over the treated area. In cases where fuel reduction was the main objective, not only was fuel reduced but fire ladder structures were interrupted. These types of burns are scheduled for fall or early winter when the heavier fuels are still dry. The areas treated for fire hazard protection were chosen according to proximity of residences and high recreation use areas.

Wildlife

Because of the highly diverse burning conditions created by both weather and fuel distribution, wildlife has received the greatest direct benefit from Edison's burn program. The results can be measured and observed in numerous ways. Deer browsing has been the most dramatic and immediate response observed. The increase in 'edge affect' has allowed for both retention of cover vegetation while creating adjacent openings for browse and forage. Fixed and project specific transects have been run since 1996 by the staff wildlife biologist. Song bird transects indicate the expected results in both richness and diversity.

Timber production

Tree growth has been enhanced in several ways with the burn program. Reduced competition is the primary means of increasing wood fiber production. Natural pine reproduction has been greatly enhanced and in most cases we have experienced significant increases in all species. In fact, in some areas we are now experiencing too much reproduction.

Water

As vegetation is reduced, water has a better chance to get into the lower levels of the soil. This greatly enhances the downstream flows later in the year. In no case has fire caused water quality degradation. Research has not proven unequivocally the amounts of increased water availability, however, any increase is important. Numerous small wet areas have appeared in spring and early summer, following burning, indicating an increase in ground water resources.

Vegetation

The vegetation found in the Sierra Nevada has been influenced by fire for nearly 15,000 years. Many species not only tolerate fire, but require fire for rejuvenation. We have overwhelming evidence to support this fact. Our burning program has not only increased the variegated nature of vegetative patterns, but has created an abundance of edge effect options by altering the patterns of openings within, and type changes adjacent to brush fields. In the wet areas mentioned above, both grasses and sedges have increased dramatically.

Soils

Soil temperatures may be a concern if the area is too extensive. While higher temperatures can do harm to organic components of soils, no adverse impacts have been observed in this program. In fact, during a field survey early in the program a small area (50 feet by 4 feet) that had extremely high temperatures was monitored. All of the organic material had been consumed leaving only bare glazed mineral soil. Thousands of microscopic lichens and fungi were later observed growing on this soil. This over heated area provided optimal conditions for these types of pioneer plant species. This observation provided further evidence that varied conditions and fire behavior create the desired mosaic which benefits all species of vegetation and wildlife as well as soil conditions. [Additional site specific information on the above categories are available from the author]

Conclusion

While our fire program has been successful by itself, mechanical manipulation has also been used to increase our ability to apply fire to the forest. We have used normal sawlog harvesting, cut-to-length, and bio-mass thinning operations, to reduce fuel loading. The greatest direct benefit, other than income, of these operations has been fuel reduction which allows for hotter burn prescriptions. This may provide a logical enhancement to the reintroduction of fire into the Sierra Nevada ecosystem on public lands. Hotter prescriptions also reduce smoke impacts, as well as reduce the need for repeat treatments.

The future of our burn program while not over, appears dim, primarily due to cost considerations. The community of Shaver Lake supports burning and now understands and puts up with smoke, knowing the beneficial results of fire. However, clean air restrictions through various laws and regulations are increasing the costs of completing a burn project. As mentioned above the benefits of burning are intrinsic and not easily measured. Therefore budgeting becomes difficult, especially for a private forest owner. Without measuring economic benefits, offsets to capital outlay cannot be justified.

The overriding objective for our program is that we know that fire is a major player in the health of the Sierra Nevada ecosystem, and without fire the balance will be further disrupted. Many of the problems with insects, disease, and reduced biodiversity stem from lack of fire over the past 100 years.

Our thrust in the future will be to continue to burn, but also to increase our monitoring program. Not only will monitoring allow for refinement of our knowledge of fire, but for evidence that fire is required and should be exempt, or at least considered separately from air quality standards and other budgetary constraints. I do not believe that we should burn whenever we want, but fire should be recognized has a natural resource with protections equivalent to other natural resources, such as old growth. One major point that needs to be clarified, is that we must burn under fire’s terms not ours. Our schedules and budget constraints often conflict with desirable burn conditions and this does not produce the desired results. There is no such thing as a bad fire unless it creates a large expanse of a singular eco-type, such as results from a catastrophic wildfire.

Use of Short Rotation Burning to Combat Non-natives and their Seed Banks in California North Coastal Prairie

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Abstract

We have been conducting and monitoring the effects of prescribed burns aimed at thwarting invasion by exotic pest plants on public land flanking Mt. Tamalpais in Marin Co., California for five years. This area is just north of the Golden Gate in central California. French broom (*Genista monspessulana*) and yellow starthistle (*Centaurea solstitialis)* are the most pernicious invaders here; both are capable of completely displacing grassland vegetation. Previous research in our grassland site suggested that stands of French broom reach a threshold at some age after which they will be extremely difficult to eliminate because of the accumulation of huge seed populations that become distributed relatively deeply in the soil. However, seed bank sampling after our most recent burn documented that seed populations in areas where old stands had occurred have plummeted since the initial inventory. Seed mortality caused directly by the 1998 fire was not detectable. Without natural seed predators, we assume predation on seed stored in the soil also contributed little if any to the decrease. Annual data on seedling emergence only explained a tiny fraction of the decrease. Apparently undetected germination was important. Age-related seed mortality may have also occurred. The percent of seed that germinated in samples from old infestations was lower than from young, possibly indicated an age-related reduction in viability, and/or a more impervious seed coat among remaining seed. Continuing to prevent input of new seed following initiation of a burning program will be crucial to maintaining the depletion trend. This prevention can be accomplished by additional burning before seedlings reach reproductive age. Fortunately, at our sites repeat burning has increased the cover of native plants and appears to have minor negative ecological consequences (coyote brush mortality, increased cover of annual grasses).

We also present data documenting how burning three consecutive years has virtually eliminated yellow starthistle at another site. This annual species does not maintain a persistent soil seed bank. Annual grasses have filled the void left by its disappearance. There has not been a detectable increase in native cover where starthistle burning has occurred.

Introduction

French broom (*Genista monspessulana*) and yellow starthistle (*Centaurea solstitialis*) are non-native pest plants that have profoundly degraded biological and economic resources in the Western U.S. (Hoshovsky 1988, Herzog and Randall 1992). Both invade grasslands in Marin Co. and elsewhere in Northern California, displacing them with near monocultures of 1-3 m tall shrubs, or herbs with spiny yellow heads. Broom also invades woodland and forest vegetation, but it is not as abundant overall as starthistle, which now covers about 10 percent of California's grassland habitat according to the California Department of Food and Agriculture's mapping. This valuable forage production is almost a total loss (Pimental et al. 2000).

Following removal both species quickly recolonize from seed. Fire may be a useful tool for combating infestations of these weeds because it can reduce or eliminate this source of regeneration. However, to do this effectively fire must be undertaken every 1-3 years to kill re-emergents before they grow up and disperse new seed. Such frequent burning deviates greatly from the historic fire regime of the native vegetation. Reconstructed mean fire intervals in the region have been found to be ~8 years (Brown et al. 1999), ~14 years (Finney 1990), between 6 and 23 years (Finney and Martin 1992), or 20-30 years (Jacobs et al. 1985) depending on location. Extreme modification of a fire regime will tend to cause dramatic changes in vegetation (Whelan 1995, Agee 1999), and, ironically, increased fire frequency may promote invasion by non-native pest plants (D'Antonio and Haubensak 1998). We have studied repeat fire treatments to help determine the effects of altering the fire regime on native vegetation.

In addition, the number of successive burns that must be undertaken to potentially achieve control of French broom and yellow starthistle is uncertain. Parker and Kersnar (1987) and Odion and Haubensak (in press) found 5,000-10,000 seeds/m^2 in the soil under French broom. Initial work at the broom study site reported on here suggested that stands reach a threshold at some age after which they will be extremely difficult to eliminate using burns because the size and depth distribution of the soil seed bank allows a high proportion of seeds to persist through fire (Odion and Haubensak, 2002). Such persistence of seed at depth through fire has been well-documented for *Acacia sauveolens*, another woody legume with fire-stimulated germination (Auld 1986). A key question for broom is: what is the decay rate of the persistent seed bank? We have now conducted three burns in four years to prevent new seed input at the site reported by Odion and Haubensak. Seed banks were assessed after the third burn to determine seed bank persistence following this number of treatments and growing seasons where both young and old infestations initially occurred. By understanding seed bank persistence, land managers can estimate the number of treatments that will be required to achieve control. This knowledge is integral to optimizing control efforts because a long term approach of repeat pre- and post-burn treatments that rotate over a landscape can maximize effectiveness (Swezy and Odion 1997). In addition, if burning is not continued long enough it can backfire and cause infestations to become more extensive (Odion and Haubensak, 2002).

Unlike French broom, yellow starthistle is an annual. Seed can live 3-8 years depending on depth of burial (Joley et al. 1990). Burning when the seeds are still in the involucres can eliminate annual seed production. With no seed input, the soil seed bank can be depleted in just a few years, due mainly to germination (Joley et al. 1990). However, there is a narrow window of time before seeds are dispersed when associated grass fuel can carry fire through still green yellow starthistle infestations. Fires at our site, with its relatively wet climate and late-curing grass, were subdued, difficult to sustain, since the fuel was too moist to be consumed. Are fires under these circumstances effective at eliminating starthistle seed?

Methods and Materials

Research was conducted on the Marin Municipal Water District Watershed lands in Marin County, California. Broom has been studied at Bon Tempe Meadow, elevation 200 m. Mean annual precipitation is about 120 cm, monthly average maximum temperature ranges from 13 ^{o}C in

December to 25 ^{o}C in the summer. Starthistle research has been conducted at Rock Spring Meadow at 650 m elevation on the southwest flank of Mt. Tamalpais. Year-round climate data are not available. Due to its elevation, this meadow receives slightly more precipitation and is slightly cooler in winter and warmer in summer compared to Bon Tempe Meadow.

The meadows are underlain by sandstone and shale of the Franciscan Assemblage. Bon Tempe meadow is almost entirely within the Blucher-Cole complex soil mapping unit (USDA Soil Conservation Service 1985). These soils are very deep clay to silt loams. Permeability is moderate to low. Soils are subject to brief periods of flooding. In the area where plots were located, slopes are 0 to 10 percent with a south aspect**.** Soils in the Rock Spring meadow are mapped as the Saurin-Bonnydoon complex. The unit averages 50 percent Saurin Clay and 30 percent Bonnydoon gravelly loam. Permeability is moderate. Slopes in the areas studied range from 2 -15 percent.

Meadow vegetation consists of grassland with patches of broom or starthistle. Individuals or clumps of coyote bush (*Baccharis pilularis*) and live oak (*Quercus agrifolia*) are also common in Bon Tempe Meadow. Grassland consists of a matrix of native coastal prairie (Holland 1986), and grassland dominated by exotic species. Native grassland is composed of fairly dense to dense tussocks of needlegrass (*Nassella pulchra*) and oatgrass (*Danthonia californica*). Non-native grassland consists of two types. In dry, loamy soil, rattlesnake grass *(Briza maxima*) is dense. Scattered with it are the annuals wild oats (*Avena barbata*), quaking grass (*Briza minor*), *Vulpia bromoides* and *Aira caryophyllea*. In wetter, low-lying areas, the robust perennials velvet grass (*Holcus lanatus*) and vernal grass (*Anthoxanthum odoratum*) occur. Nomenclature follows Hickman (1993).

Experimental Design

Broom--Layout of plots for broom research is described in detail in the Proceedings of the 1997 AFE Symposium (Odion and Haubensak, 2002). Prior to burning, 36 and 33 plots were established in relatively young and old stands of broom respectively. Plots were .5 x .5 m and located at eleven 5 x 10 m sites. Young stands had an average of 185, 1 to 2 m tall broom plants/m^2, the largest estimated to be about 5 years old based on their size and observations of rings on cut plants. There was a dense cover of grass and thatch. Old stands had an average of 88 broom plants/m^2 and the largest were estimated to be about 15 years old. Associated vegetation averaged 5 percent cover and was mainly poison oak (*Toxicodendron diversilobum*) and grassland remnants.

Before burning in October 1995, broom was cut near soil level and allowed to thoroughly dry. Slash was left in place in 1/3rd of the plots (hereafter referred to as slash plots). Slash was concentrated by a factor of 4 in 1/3rd of the plots (fuel addition) with material collected nearby. Slash was removed from 1/3 of the plots. This treatment was done with the hope of simulating the effects of long duration soil heating in chaparral fires, which has been shown to cause particularly high levels of seed mortality (Odion and Davis, 2000).

After fire in 1995, five soil cores (5 cm diameter by 9-10 cm depth) were taken at random locations from within fuel-added and fuel-removed plots. The distinctive seeds of French broom were then retrieved from 1,000 cc homogenized subsamples. These were inspected under a microscope and classified as alive (white, undamaged embryo) or dead (brown, black, or charred). The total alive and dead was used as the estimate of the pre-burn seed bank. Because of the level of sampling, this should be considered a rough estimate. A subsample of 150 live seeds was spread on wet paper outdoors to evaluate germination.

Burns were again carried out in the study area in mid-October in 1996 and 1998. Before and after the 1998 fire, three of the same-sized cores were collected from each plot. These samples were spread in a thin layer over potting soil in growing trays. A light dusting of potting soil was sprinkled over the natural soil layer. Trays were irrigated regularly and maintained outdoors in a lathhouse on the UC Berkeley Campus. Broom seedlings were counted as they emerged. After germination, seeds were retrieved by sieving. Viability of ungerminated seeds is currently being assessed. This more detailed seed bank inventory should provide a good estimate of remaining seed.

Starthistle--Six patches of yellow starthistle were identified in Rock Spring meadow prior to burning. One transect was randomly located within each. Transects were 10 m long. We measured point intercept cover every .25 m along each transect during late spring each year following fire. The meadow was burned each year from 1996-1998 in early to mid July, just as grasses were beginning to cure. Depth of ground char was visually characterized at each point along each transect following fire (Ryan and Noste 1985).

Data Analysis

Paired t-test were performed comparing plot and transect percent coverages in different years. Percent cover data were arcsin transformed.

Burn Characteristics

Temperatures and relative humidities ranged from 25 to 28 ^{o}C and 30 to 35 percent respectively when broom plots were burned. In young stands where there was abundant grass fuel each year, flame lengths were 0.5-1.5 m, and ground char was classified as light, or moderate where broom slash was most abundant (first burn only), including all plots where fuel was added. Fuel addition plots had residence times of 4 to 5 minutes for flaming combustion and flame lengths of 3 to 4 m. Subsequent burns all had relatively high fireline intensity for grassland types in California, although ground charring was classified as light. Old broom, slash plots (year 1) had up to 15 cm of broom slash that produced 0.5 to 2.5 m flame lengths that lasted 2 to 4 minutes. Soils were classified as having moderate ground char over 80 percent of their surface and deep ground char over the remainder. In fuel addition plots, residence time of flaming combustion was 4 to 6 minutes and smoldering combustion continued long after. All plots were characterized as having deep ground char. Fuel present at these sites and in year two consisted mostly of broom seedlings with sparse grass. Combustion during the second burn was light, and the soil surface was unburned in 90 percent of plots and lightly burned in the other 10 percent. By the third burn, annual grasses had colonized all plots and fire behavior was typical for sparse grass fuel. Ground charring was light.

Starthistle burns were low intensity grassland fires, with flame lengths averaging about .25 m. Fires failed to consume some of the forbs, including yellow starthistle. However, these were damaged by scorching and wilted soon after fire. The percentage of line-intercept points classified as unburned was 7.7, 5.3, and 4.9 for the three burns respectively. Ground char was classified as light at all remaining points, as there was no woody fuel present.

Results and discussion

To simplify the following explanation, data from plots located where there were initially old or young stands of broom that were eliminated with the first burn will be referred to as data from old or young stands even if they were collected after broom infestations ceased to exist.

Response of French broom Seed Banks to Burning--Remarkably, seed populations decreased an order of magnitude in old stands *following* the mortality induced by the smoldering combustion of slash during the initial burn (Table 1). We found no differences in the 1998 seed bank between fuel added and slash plots, so data for these are combined in Table 1. Fuel removal plots are excluded from the analysis of old stands because seed populations after the first burn were so much greater there. The maximum remaining seed bank in Table 1 is the number of seeds remaining after the first burn minus all subsequent emergence measured each year in mid-winter.

Table 1. Seed and seedling density (per m^2) in broom invaded prairie. Estimated remaining seed bank in 1998 is based on the initial post-burn seed bank minus subsequent germination.

	old stand	**young stand**	**fuel addition (young)**
pre-burn seed bank	8600 ± 3900	1638 ± 1463	1638 ± 1463
post-burn seed bank	5934 ± 2670	1495 ± 1335	1441 ± 1287
1996 post-burn emergence	1650 ± 125	317.2 ± 107.6	706.8 ± 220.0
1997 post-burn emergence	368 ± 108	126.8 ± 49.3	13.6 ± 11.2
1998 emergence	1.2 ± 0.6	11.5 ± 4.5	0.8 ± 0.4
maximum remaining seed bank	3914 + 2436	1039 ± 924	720 ± 642
actual seed bank	488 ± 377	1037 ± 1344	378 ± 281
1999 post-burn emergence	0.4 ± 0.7	1.6 ± 0.5	0.6 ± 0.4

The 1998 pre- and post-burn seed bank samples had equivalent densities. With germination or death of surface seed following previous burns, remaining seed are mostly low enough in the soil to survive soil heating produced by combustion of the relatively sparse annual grass. In fact, the germination rate in the lathhouse also did not differ between pre- and post-burn samples (Alexander, unpublished data), so apparently seeds did not receive enough heat to affect seed coats. Seed mortality in the 1996 burn was presumably nil as well in old stands since most plots there were classified as unburned. Thus, about 3500 seeds/m^2 disappeared that are unaccounted for in old stands (Table 1). With natural post-dispersal seed predators lacking, predation on seed stored in the soil probably contributed little if any to this.

For Scottish broom, Sheppard et al. *in press* with detailed monitoring of emergence, were able to attribute the 36 percent seed bank decay rate they measured to germination. Undoubtedly there was germination in our old stands that was undetected due to early seedling mortality and/or failed or late emergence, especially after the first burn, when a huge flush of germination occurred. A previous experiment (Odion and Haubensak, 2002) found that bringing seeds closer to the soil surface by turning over soils doubled the subsequent emergence compared to controls. This is not likely due to seed remaining dormant in controls, as broom would be expected to germinate readily with sufficient moisture once the innate physical dormancy imposed by its impermeable seed coat is overcome. For Scottish broom, Bossard (1993) reported that temperatures did not provide any barriers to germination. Cold stratification was not needed as a pre-treatment in germination studies on French broom (Parker and Kersnar 1989, Adams and Simmons 1991). Thus, the experimental results may have resulted from failed emergence of seedlings from the more deeply buried seed in control plots. Based on the average weight of broom seeds (6.3 $\pm$ 1.1 mg for 15 seeds retrieved from the soil in old stands) seedlings may only be able to emerge from depths of about 5-6 cm (see Bond and van Wilgen 1995, Fig. 5.12).

Considering how rapid the depletion of the seed bank in old stands has occurred, age-related death of seed may have also been a factor. We are not aware of any data on survivorship of French broom seed over time. For many species, seed maintained under storage will have survivorship with time that fits a negative exponential distribution (Baskin and Baskin 1999, Figure 7.8). The rapid decrease in broom seed populations in old stands might be because a portion of the seed that remained there after the first fire and germination event was old enough to be past the onset of age-related mortality. French broom seeds are prone to fungal attack (Parker and Kersnar 1989). There is additional evidence that seeds in old stands may have decreased viability. The emergence of seedlings from 1998 pre- and post-burn samples (presumably = germination because they were not deeply buried) was found to be only ~12 percent, whereas it was ~38 percent in samples from young stands ($P < .05$, arcsin transformed data). A more impervious seed coat on average among remaining seed in old stands could also explain the discrepancy. Germination behavior of French broom seed has been described as polymorphic because some seed germinates readily, while some remains dormant (Adams and Simmons 1991). Perhaps there is also polymorphism in the extent of dormancy as a result of variation in seed coat thickness or other factors that contribute to maintaining the physical dormancy.

The decrease in seed populations in old stands subsequent to the first burn and germination event is illustrated in Figure 1. If the decrease is assumed to be linear, and maintained by preventing

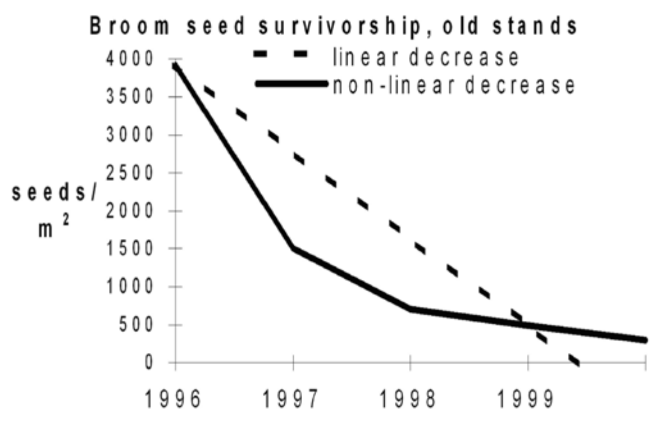

Figure 1. Decrease in French broom seed populations in old stands after the initial burn and germination event.

any new seed input, then broom seed would be gone very soon. However, this optimistic scenario is not likely for two reasons. 1.) Undetected germination in year 1 would reduce the y-intercept and flatten the slope of the line. 2) The decrease may non-linear and asymptotic (Figure 1), as has been documented for yellow starthistle seed in the soil (Joley 1990). Remaining seed in the soil in old stands may be especially persistent if they have not lost viability because such a high percentage will remain dormant.

The situation in young stands is unclear. Seed densities in our initial samples varied from 600 to 4400/m^2, so the depletion rate cannot really be determined. Our results do indicate the seed bank is initially more persistent in young vs. old stands (Table 1). This could be because age-related mortality has not been a factor. Based on the aforementioned germination results, annual germination should continue to deplete seed banks in young stands to a greater extent than old. Further, there was greater germination in samples collected after the 1998 fire compared to before. With more surface seed because of better survival with the first burn, and relatively lush grass growth in young stands, fire will facilitate seed bank depletion there. However, the seed banks in young stands should gradually become more like those in old stands over time. Another difference between young and old stands is that the fuel addition treatment in young stands manifested lower seed populations as a result of both mortality during the initial fire, and greater germination subsequently (Table 1).

More detailed analysis of French broom seed banks involving additional sites is being prepared by Janice Alexander and will be available with the completion of her Master's thesis at UC Berkeley. With an additional sampling at Bon Tempe Meadow, it should be possible to predict with more confidence when seed populations might be depleted. However, the present uncertainty should not diminish a conclusion that can be made from the data currently available—three burns and 4 years of time with no seed input have eliminated all but a tiny fraction of an enormous seed bank. Further, in contrast to the previous pessimistic assessment (Odion and Haubensak, 2002), it appears that long-established stands of broom can be controlled with a finite number of repeat burns. Continuing to prevent input of new seed will be crucial to maintaining the trend of seed bank depletion. This prevention can be accomplished by additional burning before the current crop of seedlings reaches reproductive age.

Changes in plant cover with repeat burning--Broom that resprouted after the first and second burns were all killed by the third burn. The only remaining foliar cover of broom after three burns was from the few seedlings that emerged in 1999 and 2000. The decrease in broom cover in young stands was accompanied by a significant increase in both other non-native species and native species (Figure 2). Non-native species were annual grasses, including (in order of 1999 abundance) *Avena barbata*, *Lolium multiflorum*, *Briza maxima*, and *Aira caryophyllea*. Conversely, the native species were mostly perennial grasses (*Nassella pulchra, Danthonia californica, Lemus triticoides*) and forbs (*Chlorogalum pomeridianu*m and *Sisyrinchium bellum*). *Nassella pulchra* was expected to respond favorably to repeat burning based on previous research (Ahmed 1983). Because other species had been all but choked out by French broom in old stands prior to treatment, burning had a greater effect (Figure 3). However, in old stands where fuel had been added, native cover only increased to 7 percent, significantly lower than where slash was left in place for the year one burn (Figure 3). Native species contributing to the increase in cover in old stands were primarily the grasses

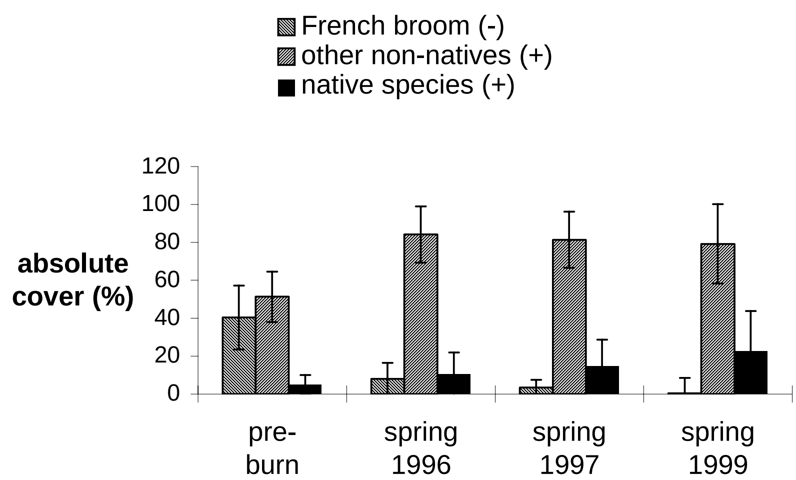

Figure. 2. Percent cover of French broom, other non-natives, and native species in young broom plots, n = 36. No difference was found among treatments for species other than broom, therefore, plots from all three treatments were combined for analysis of burning effects on these plants. A plus or minus following a legend entry indicates a significant increase or decrease in cover for that entry over the length of the study. Stars below the x-axis indicate a significant difference in cover of other non-natives or natives.

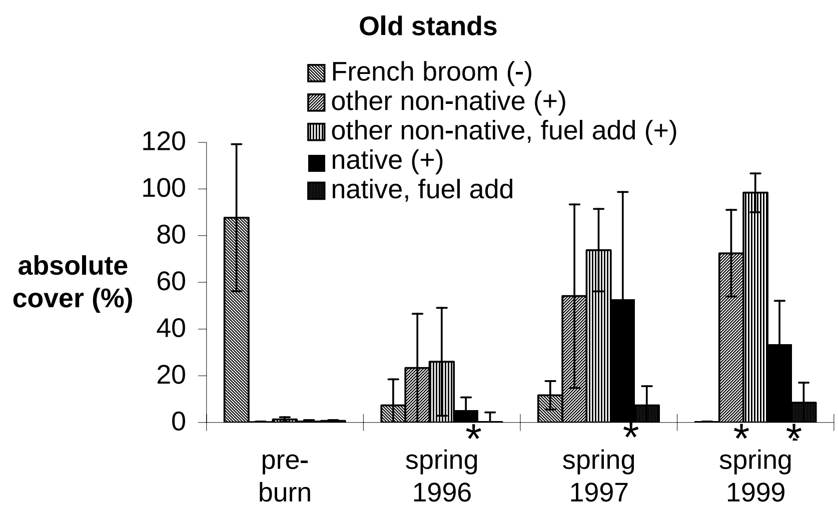

Figure. 3. Percent cover of French broom and other non-natives and native species in old broom plots. Plots where fuel was added prior to the first burn (n = 12) are shown separately from those in which slash was not manipulated (n = 9). A plus or minus following a legend entry indicates a significant increase or decrease in cover for that entry over the length of the study. Stars below the x-axis indicate a significant difference in cover of other non-natives or natives between slash and fuel add plots.

Nassella pulchra and *Danthonia californica*. In addition to slowing native species recovery in old stands, the fuel addition treatment resulted in significantly greater cover of non-natives by spring, 1999. Interestingly, the effects of the fuel treatment in year one became increasingly pronounced over time even though all plots were subjected to the same treatments after year one.

We conclude that fire is an effective tool for the control of French broom if undertaken on a repeat basis to prevent reemerging plants from reaching reproductive age, and if done several times. It is possible to reduce seed populations to low levels while eliminating seedlings and resprouts. In addition, we could document no negative effects of fire on native species, instead finding modest increases. It should be pointed out that other pernicious non-natives that qualitatively appear to thrive with fire, velvet grass (*Holcus lanatus*) and vernal grass (*Anthoxanthum odoratum*) were not common in the locations where plots were established. In addition, repeat burning did all but eliminate one native species that was not in our plots, coyote brush (*Baccharis pilularis*).

Starthistle

Yellow starthistle was intercepted at more than half of the 600 points prior to burning in 1996. After three burns, it only occurred at six points (Figure. 4). The six points were not among

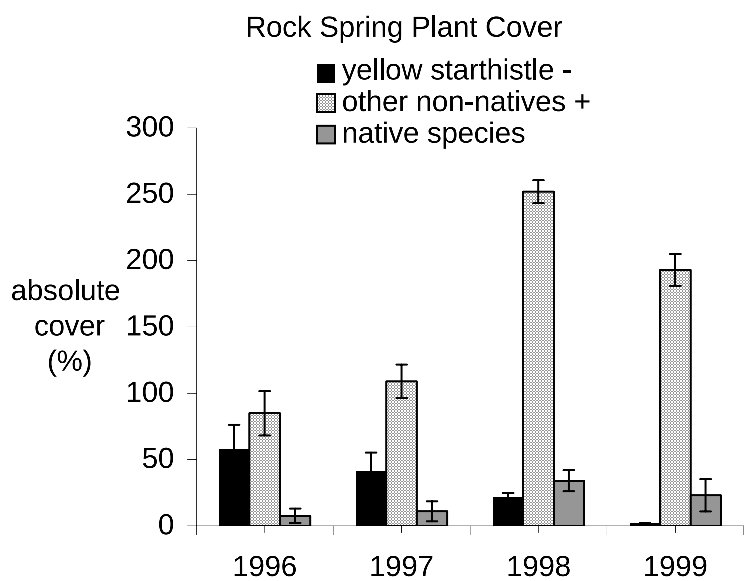

Figure. 4. Changes in absolute cover of yellow starthistle, other non-natives, and native species from prior to the first burn in 1996 to spring 1999, after 3 burns. A plus or minus following a legend entry indicates a significant increase or decrease in cover for that entry over the length of the study.

those classified as unburned following any of the burns. Despite the precipitous starthistle decline, at least one more burn will be required to eliminate the infestation. There was a significant increase in other non-native species with burning. There was no difference in native species cover ($P = .08$). The increase in non-native cover was due to annual grasses (Figure 5). (*Avena barbata*), rattlesnake grass (*Briza maxima*), a fescue (*Vulpia bromoides*) and *Aira caryophyllea* all increased dominance

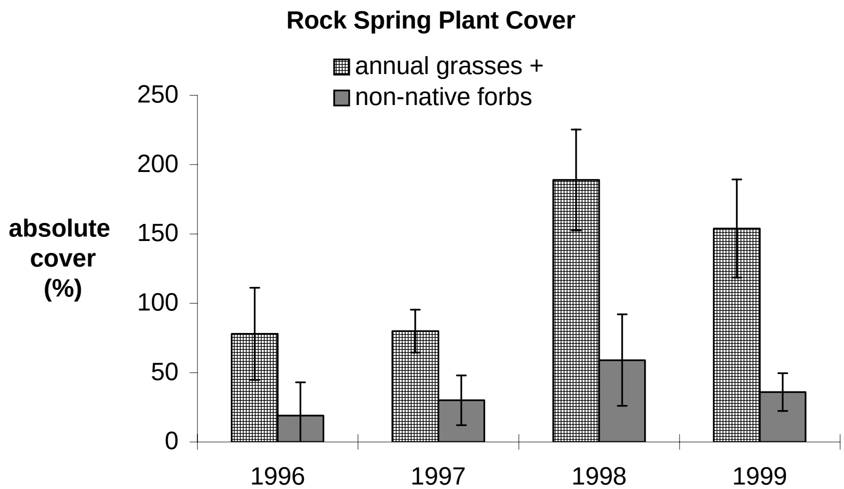

Figure 5. Changes in absolute cover of annual grasses and non-native forbs from prior to the first burn in 1996 to spring 1999, after 3 burns. A plus following a legend entry indicates a significant increase or decrease in cover for that entry over the length of the study.

with burning. Cover of non-native forbs (mostly other thistles and *Geranium molle*) remained unchanged (Figure 5). Native herbs (clovers, *Trifolium* spp., blue-eyed grass *Sisyrinchium bellum*, and *Brodiaea* spp.) and grasses (*Nassella* and *Danthonia*) both appeared to increase modestly over the course of the study, but no significant increases were detected (Figure 6). The mean number of

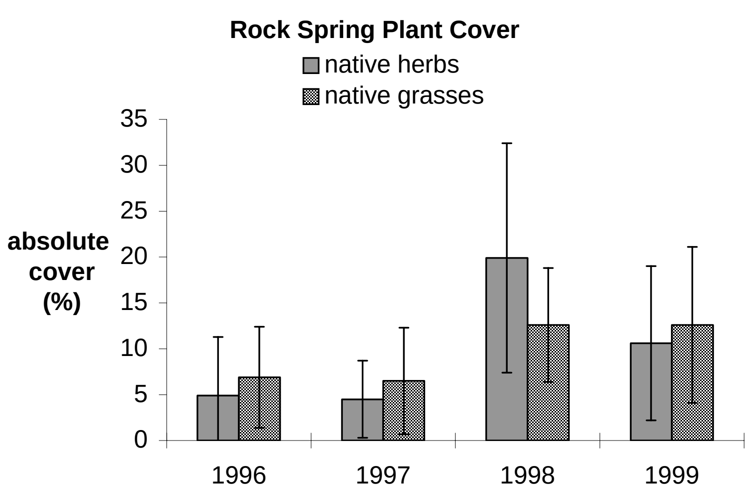

Figure 6. Changes in absolute cover of native herbs and native grasses from prior to the first burn in 1996 to spring 1999, after 3 burns.

species per transect was greater in 1999 compared to 1996 (Figure 7). This was the result of a significant increase in the richness of non-natives.

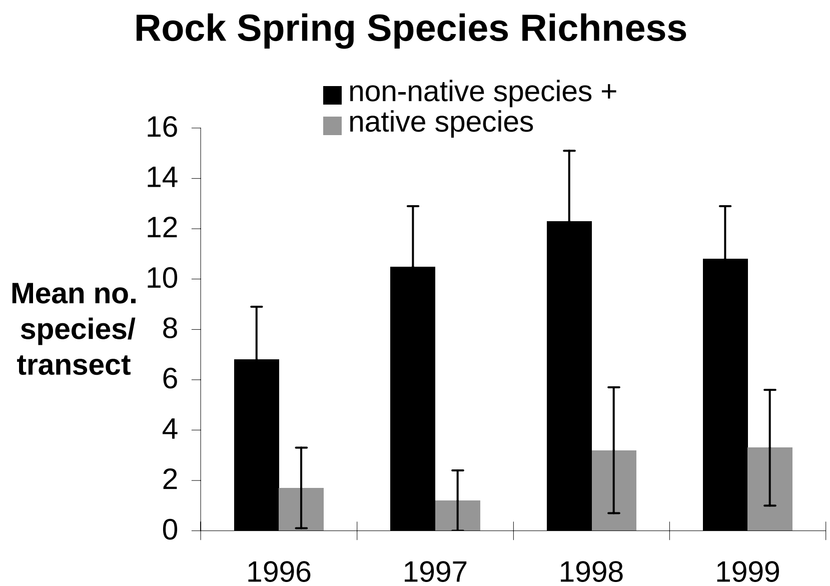

Figure 7. Changes in numbers of non-native and native species from prior to the first burn in 1996 to spring 1999, after 3 burns. A plus following a legend entry indicates a significant increase for that entry over the length of the study.

Although at least one more burn will be required to eliminate the starthistle infestation, annual burning is an excellent tool for starthistle control. Yellow starthistle may be replaced with a suite on non-native grasses, but these at least provided forage value and food chain support. Further, the annual burning did not have detrimental effects to native species that we could document, and it is possible that natives did actually benefit.

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Smoke Management: A Practical Approach to Addressing Health Hazards, Safety, Visibility, and Costs

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Abstract

In the United States a successful "burning is bad" campaign, implemented over the last 90 to 100 years, has lead to wild-land and forest areas that are unhealthy and overgrown. This campaign has also created dangerously high biomass loadings, which routinely fuel the flames of devastating wildfires. It has become critical to reestablish a more natural balance in the forest ecosystems and to reduce biomass loadings for public safety and welfare. Land stewards maintain that the key to improving forest health and reducing wildfire threat is the reintroduction of fire into our forests through more extensive prescribed burning activities. However, prescribed burning means fire and flames in our forests and this is an unpopular concept with the public. Where there is fire, there is also smoke, and smoke contains many harmful air pollutants. Because of this, air quality professionals and public health experts also find the idea of increased prescribed burning difficult to embrace. This is understandable. However, there are some practical approaches to bridging this gap. For example, addressing the social, political and economic impacts of prescribed burning. There are ways to minimize the generation of smoke, minimize human exposure to such smoke, and minimize or eliminate smoke impacts on sensitive members of our populations. There are also ways to minimize impacts on visibility and handle the costs associated with these efforts. A simple, practical model for addressing these issues is presented.

Keywords: Smoke Management, Prescribed Fire, Practical Communication Model

Introduction

In the United States a successful "burning is bad" campaign, implemented over the last 90 to 100 years, has lead to wild-land and forest areas that are overgrown and unhealthy. Fire suppression, although well meant, has reduced bio-diversity, stressed ecosystems, and supported plant and animal pest infestation. It has affected wildlife, wildlife habitats, forestry, recreation, public trust and safety. Consistent and widespread fire suppression has contributed to greater biomass fuel loadings. It has created many acres of land with dangerously high ladder fuel volumes, which now feed the flames of characteristically devastating wildfires that threaten life and property. Wildfires are becoming larger and hotter. They are lasting longer and cause more damage than ever before. These same wildfires create serious economic hardship on every level—local, state and federal. Most land managers agree that the majority of forests will burn, although it is difficult to determine when, with what intensity, and for how long (Gascoyne 1996).

It has been suggested that the key to improving forest health and the solution to reducing the threat to life and property of the many thousands of people who live in wildfire prone wildland/urban interface areas of the nation, is the reintroduction of low- to moderate-intensity prescribed burning (California Board of Forestry 1997, Mutch 1994). However, the re-establishment of healthy forests and increased public safety through the use of prescribed fire, although simple in concept, is complex and very challenging to implement.

The increased use of prescribed fire in forested areas is a largely unpopular concept with many people, including other environmental resource managers. Where there is fire there is smoke, and smoke contains many harmful air pollutants (Jenkins 1996, Black 1997). This is a great concern to air quality and public health professionals. Smoke also degrades visibility. Poor visibility can impact recreation and tourism. This in turn impacts local economies and is a concern to local governments. However, the largest source of popular opposition to the use of prescribed fire is the public.

Public Psychology

"Smokey the Bear" did a superior job of teaching us that "Fire is Bad." Out of all of us who pay taxes and take the time to vote, who doesn't know that "Only you can prevent forest fires"? I still on occasion have nightmares about Bambi in the forest, searching desperately for his mother amidst a sea of flames, smoke and disaster. Couple these images with those of red hot flames and choking walls of smoke from wildfires, which end up on TV and on the front page of major newspapers and nationwide magazines during every wildfire season, and you have a public psychology that will naturally associate fire with "bad things".

Interestingly enough, the key to addressing the numerous land management resource issues, public safety and economic challenges—the "bad things" that we associate with catastrophic fire—is indeed, in many cases, the reintroduction of controlled fire into the forests. To be successful in this effort, to free up the resources needed and to create the viable sustaining programs that will provide the needed solutions, the public psychology that associates fire with "bad things" will need to be addressed. In this case it is helpful to remember that other resource agency professionals, politicians, public health specialists, and wildland stewards, having grown up with or otherwise implemented the original "Smokey the Bear" campaign, also share this same public psychology. Because of this, the public psychology will have to be modified before prescribed fire will be embraced at any level. It will have to be expanded to incorporate two related dimensions: 1) associating devastating catastrophic uncontrolled fire with "bad things" like damaged property, loss of life, and serious economic impacts, and 2) associating well-controlled, deliberate, low intensity prescribed fire with "good things" for all of us. Making known what the good things are, and how they are of value to all of us is a critical component of this effort. So is getting the public to understand and believe in the science behind why moderate to low intensity fire is important and how it will and won't be used. A well-coordinated and consistently implemented education, outreach and communication effort is the only effective way to approach this.

A "Smokey the Bear" type campaign could work very well here, as it did in the past, although "Smokey's" message would have to be revisited. "Smokey" could put forth the idea that "uncontrolled wild fire" is "bad" while "controlled prescribed fire" is "good" in some cases. The concept itself is not too difficult to grasp. There are many obvious examples where the public clearly understands the concept of "too much is bad, but a little can be good". For example, the public is well aware that nuclear radiation in high does causes cancer and can kill, but they also know that in controlled, low dose settings, radiation helps cure cancer. The public can easily fathom that sunlight (UV Light) in large doses can be harmful by aging our skin prematurely or by causing skin cancer, but in moderate doses it makes us feel good and it is essential for life. The public has even grasped the idea that a single daily multi-vitamin is good, but that high doses of vitamins or any other nutritional supplement can be toxic. And, the public knows all to well that blatant alcoholism is bad, but that a single glass of wine a day may be good for your heart as well as your disposition.

Fortunately, when it comes to catastrophic wildfires, the public is already aware of what the "bad things" are and it already highly values the "good things", which are presently threatened. These include clean water and air, beautiful streams, lakes and rivers, healthy animals, and plants, beautiful places to walk or hike, to hunt and fish, to harvest wood or plants from, or simply to see and enjoy. In many cases they just don't realize that these good things result from the well managed balance between nature's needs and the needs of people moving to the ever dwindling and considerably fragile wild-land/urban interfaces of our forests and woodlands.

In other cases the public is aware, but has simply forgotten. During the sixties and early seventies, there was an age of increased awareness and commitment to the environment. However it happened "a long time ago". What had happened to those who made it happen? We've grown up and we're busy living our everyday lives. We have forgotten that increased bio-diversity, healthy forests and balanced ecosystems in part make our lives possible. We have forgotten that the good things don't happen magically, that they only happen when we work together, when a well-managed balance is reached between nature's needs and the needs of people. We are those people, the ones who want serenity by procuring space and distance from the busy lives we have created. We do this by moving to the wild-land/urban interfaces of our forests and woodlands where we unwittingly become part of the complex problem. However, in all cases we can easily be reminded or enlightened by those who are already aware and are willing to make the effort, to put in the time it takes to do so.

In order to bring about the reintroduction of fire into our forests awareness of the forest health and fire danger problem has to be increased. We need to be made aware of how we, the public, contribute to the problem, as well as how we, the public, will help solve it. However, there is often the expectation that public agencies, which are charged with the task of managing our natural resources as well a protecting public health and welfare, will somehow be able to do this all by themselves. It is important to remember that we have created a government of the people, for the people and most importantly by the people. While public agencies have statutory responsibilities and may take the lead on such efforts, the key to the success in addressing this problem is getting the public involved. This can be accomplished by increasing awareness and consequently participation, through a well coordinated, consistently implemented education, outreach and communication effort.

A practical model of how this works is shown in figure 1. If you create awareness of the problem you are trying to solve, and do a good job of informing, involving, and educating people, people will become invested in finding, creating and supporting solutions to the problem. Invested people necessitate public support. Political support usually follows closely on the heals of public support. If a cause has political support, then someone, somewhere will find the money to support the efforts needed to actively address the problem. This provides an avenue as well as the resources for creating viable sustaining programs. Viable sustaining programs ultimately provide the tools needed to develop and implement solutions. The key to success in this model is creating awareness. It has been done before. Just look at what "Smokey" did for wildfire fighting and prevention efforts. It can be done again. Especially since the age of electronics has given us so many more communication tools, opportunities and avenues.

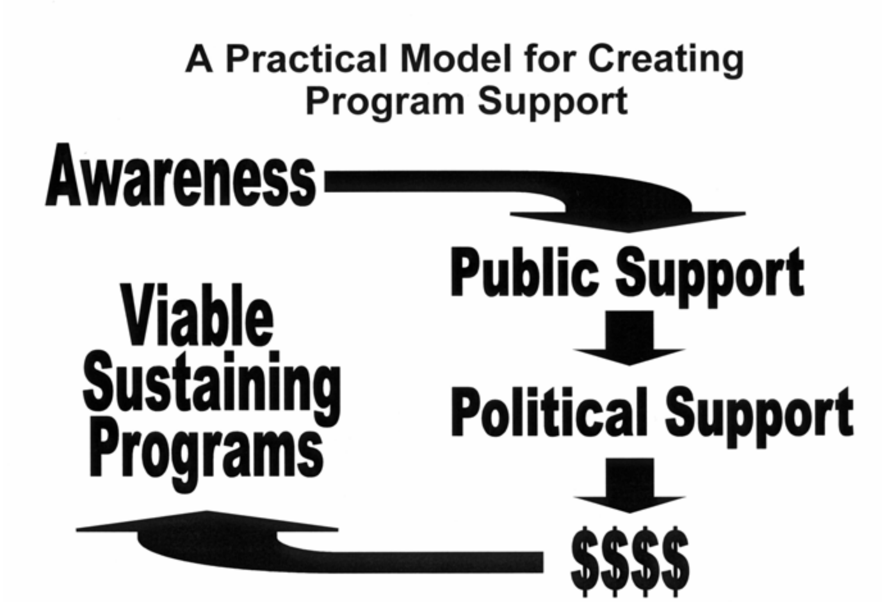

Figure 1: A simple conceptual model for increasing public support and encouraging the development of viable sustaining programs, which are needed to identify and implement solutions to the problems of decreased forest health and increased threat to public safety and welfare from catastrophic wildfire, in the forested and wildland/urban interface areas of the United States.

The Solution

While the solution seems simple and straightforward, its implementation is not easy. Because of this difficulty, the reintroduction of fire into our forest ecosystems is slow. The reason for this may surprise you. Although education, outreach and communication efforts such as the one proposed here take time to implement, this is not the reason why the reintroduction of fire into our ecosystems is not happening quickly. The reason is not that the public isn't capable of being educated or isn't willing to adapt and change its perspective, attitudes and actions based on more information and increased awareness. It isn't that the public is too wary or cynical either. Nor is it that the public couldn't be protected from the potentially harmful effects of smoke or that local economies couldn't survive managed periods of poor visibility. The reason lies within the government resource agencies themselves, within those who are charged with managing and protecting forest resources, public health, welfare and safety.

A much greater challenge than the task of changing public psychology is the task of creating unity and a common purpose between numerous government agencies. Many stakeholders are involved in the forest health and prescribed fire issue, both public and private. This means that numerous egos, attitudes and agendas, both public and private, are also involved. The fundamental challenge lies in creating awareness among agencies and in getting the many competing local, state, federal and private land managers to work together. To be successful in addressing forest health and prescribed fire issue, land management and environmental resource agencies need to find common ground. They need to first develop an internal sense of community and then go on to foster external public relationships. The simple practical model presented in figure 1 offers a place to start. This model will work at all levels in the communication effort. With a few modifications it can be used with equal success to increase awareness and build support within agencies and between agencies as well as between agencies and the public.

To be successful on the broad scale every agency involved must stop thinking that they can do it on their own, with their own resources, and should start asking for help. Agencies will have to stop competing for limited existing resources and start going after new, as yet untapped, resources. They will have to stop blaming others for creating the situation in the first place and for the general lack of progress in addressing the problem. They will have to get creative and start thinking outside the box. They need to stop thinking there is only one solution. To effectively and efficiently address an issue of this complexity, the many invested land management and environmental resource agencies will have to work together to create a common message in a language that is readily understood by the public.

The "Smokey the Bear" campaign worked well before and was widely accepted by members of the public and private industry alike because Smokey had a common, believable easily understandable message. Because it was well coordinated, organized, and consistently implemented, the campaign had value. Most importantly, the campaign was widely promoted and supported by all levels of government—local, state and federal. Because the campaign was consistently implemented year after year by many different authorities, the public not only became educated and aware, they became invested. As a consequence, they became involved. Smokey's message was shared in schools and at private functions, such as Boy Scout and Brownie meetings. It was in cartoon and in grocery stores. Because "Smokey's" message was simple and it was understandable, it was remembered.

Conclusion

In the end a shared vision will have to be created between involved agencies, common goals will have to be identified, and an easily understood, believable message will emerge. If it does, and if it is supported by all levels of government, and is consistently delivered, the public will listen. Only then can those charged with managing and protecting the forest resources be effective in modifying the public's perception of the use of fire in our forests and woodlands. Only then will unwitting damage caused by 90 to 100 years of active fire suppression begin to be reversed.

Although the forest health and use of prescribed fire issue is complex and challenging, it can be addressed. After all, this government—the very same one that has been labeled a challenge here—has put people on the moon. We are an institution of the people, for the people and by the people. If we, the people, can put men and women on the moon, we can address the use of prescribed fire to improve forest health, protect human welfare and provide public safety.

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Fire Effects on Oak Recruitment in a Southern California Woodland

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Abstract

Two spring prescribed burns were monitored in an effort to quantify the effect of fire on seedlings and saplings of *Quercus engelmannii* (Engelmann oak). Prior to the fires, 2950 individuals in both of the burn units and adjacent control areas were measured and had canopy location recorded. Immediately following the burns, plots were re-surveyed and fire damage was assessed for all individuals. Plots were re-sampled after seven months to quantify survival and the ability of individuals to resprout. The majority of seedlings (24%) were killed by fire but most saplings survived (77%). Survival and recovery of seedlings and saplings was influenced by the amount of fire damage, pre-fire size, and to a lesser extent, the site and canopy location. In an experiment conducted within the burn units, fuel loads (biomass) were manipulated around groups of seedlings to investigate the effects of fire intensity on seedling survivorship. Seedlings in plots with additional fuel had modest increases in fire damage but this did not lead to reduced seedling survival. Fire damage and seedling survival was highly variable across the experimental block suggesting large natural heterogeneity in fuel loads and thus the resulting fire intensities. In the final experiment, the effects of biomass removal and herbivory on the emergence of new seedlings was investigated. Viable *Quercus engelmannii* acorns were planted in a factorial experiment with three levels of biomass removal (burned, mowed, and control) crossed with two levels of herbivory (protected and unprotected). Biomass removal by either mowing or burning resulted in increased seedling recruitment from acorns. The mowing treatment had the greatest positive effect on emergence. The herbivore exclosures failed to deter pocket gophers. Taken together, these studies demonstrate that fire is a major factor affecting *Q. englemannii* recruitment. However, it may be difficult to implement fire as a management tool to enhance Engelmann oak woodlands since recruitment is a complex function of local fire intensity, individual plant size and other environmental factors.

Introduction

California oak woodlands are important natural resources, providing essential functions for both humans and wildlife. These Oak woodlands furnish 331 vertebrate species with habitat important to their survival; the largest number of vertebrate species recorded to use any habitat type in California. The canopies of oak woodlands serve important watershed functions by reducing runoff, increasing water infiltration and improving water quality. They also providing aesthetically pleasing landscapes, areas for recreation and are prime areas for livestock grazing (Standiford et al. 1997).

Oak woodlands throughout California appear to have insufficient regeneration to maintain current population levels (Mensing 1992, Muick & Bartolome 1987). Extensive clearing and manipulation has occurred simultaneously with poor regeneration. An estimated half million hectares of hardwood rangelands have been cleared since 1945 for urbanization and agriculture (Standiford et al. 1997). The majority of remaining oak woodlands and their ecological processes have been modified by human activity. Livestock grazing, introduced plant species, changes in native herbivore densities, and changes in the fire regime have all had negative impacts on oak survival and regeneration.

Barriers to Recruitment

Successful oak recruitment in the rangelands of California is a complex process dependent on grazing, competition, herbivory and fire. The long history of cattle ranching on California's oak woodlands has often been implicated in reducing oak regeneration (Lathrop et al. 1991a, Mensing 1992). Cattle are responsible for direct mortality of small oaks as well as degrading environmental conditions for oak regeneration. In southern California oak savannas, cattle have been found to cause greater than 50% reductions in the survival of *Q. engelmannii* seedlings (Lathrop & Osborne 1990).

In conjunction with livestock grazing came the widespread introduction of non-native invasive herbaceous flora. Soil disturbance and localized increases in nutrients caused by the presence of livestock have residual effects on fallow rangelands that favor introduced grasses and forbs over native species. Without the pressure of livestock grazing, non-native annual species are able to dominate areas due to dense efficient root systems, large seed production and early germination. These annual grasses and forbs, primarily from the Mediterranean Basin, now occupy huge areas of California's grassland that may have once been prime areas for oak recruitment. Establishment of new oak seedling cohorts can actually be lower in areas following the removal of cattle due to higher densities of non-native annual grasses (Duncan & Clawson 1980).

Invasive annual grasses are in direct competition with oak seedling and saplings for space, nutrients, light, and most importantly for soil moisture. *Q. douglasii* (blue oak) seedlings grown in the presence of two invasive grasses (*Bromus diandrus* and *B.mollis*) had consistently lower growth and survival rates than seedlings grown without competitors. Soil water potential was negatively correlated with the density of the invasive competitors. Reduced soil moisture was speculated to be the factor limiting seedling growth and survival in the presence of these competitors (Gordon & Rice 1993).

Invasive non-native annual grasses also modify the fire cycle of California oak woodlands. Non-native annual grasses produce a dense contiguous layer of biomass annually. By late spring these grasses die back and lose moisture rapidly. The result is a continuous layer of flashy fuels. This along with increased ignition sources from humans has increased the fire frequency in many oak woodlands. Grass-layer fires are responsible for direct mortality and restricting growth of seedlings and saplings (Scholes & Archer 1997). In general, short fire return intervals reduce the density of woody species (Zedler et al. 1983, Minnich 1997) and fire return intervals less than five years are known to reduce the density of oak saplings (Swiecki & Berhardt 1998).

At the opposite extreme is the effect of human fire suppression activities. Fire suppression increases the time between fire events. With greater amounts of time between fires, larger fuel loads are able to accumulate. These increased fuel loads lead too hotter and more intense fires causing greater damage to oaks of all age classes.

Human manipulations of the environment have altered the community structure of native herbivores. Increases in the density of herbivores can result from increased rates of disturbance as a result of grazing, fire, discing or dry farming. Water importation leads to greater access to a key limiting resource. The introduction of palatable weedy plant species that grow at high densities can also elevate population sizes (Snow 1972). Native herbivore populations have also benefited due to reductions in predator densities for sport and to protect livestock. Native vertebrate herbivores can substantially reduce oak recruitment. Mule deer, acorn woodpeckers, ground squirrels and scrub jays can all cache or consume huge quantities of acorns (Menke & Fry 1980, Koenig 1980, Borchert et al. 1989, Snow 1972). Additional mortality results from herbivory of seedlings and saplings by mule deer, rabbits and pocket gopher. As a result it has been found that oak seedling and sapling survival can be significantly increased with the use of protective screening (Adams et al. 1992).

Quercus engelmannii

Q. engelmannii is one of California's rarest oaks with the smallest range of any oak tree in the state. *Q. engelmannii* is a member of the white oak group that is semi-deciduous and has a relatively open canopy (Scott 1990). Engelmann oaks occur in southern oak woodlands in a continuum of habitats from riparian woodland to savanna. Their range is from southern Los Angeles County to northern Baja California (Lathrop et al. 1991b). Grassland or open coastal sage scrub dominate the understory of Engelmann oak woodlands.

Physiological adaptations of *Q. engelmannii* to xeric conditions allow Engelmann oaks to establish and survive through the harsh summer droughts of the Mediterranean climate. The acorns of *Q. engelmannii* have high initial moisture content, do not require additional water uptake prior to germination, have a self-rooting mechanism and have virtually no dormancy (Snow 1972). These factors allow for rapid germination of acorns after falling off the tree even in the absence of additional available moisture. Early germination allows Engelmann oak seedlings to attain greater root biomass and rooting depth before winter rains have ceased. Under unfavorable conditions, developing seedlings can delay shoot development for up to 3 years. Delayed shoot development allows for greater root growth prior to the stress imposed by transpiring leaves (Lathrop & Osborne 1990). Seedlings that have already sent up shoots cope with moisture stress by dropping their leaves. *Q. engelmannii* seedlings also resprout readily from shoot death following desiccation and fire (Snow 1972).

Following germination, Engelmann oak seedlings require at least partial shade to survive. Seedlings survive best under the shaded canopy of trees and shrubs (Griggs 1987, Lathrop & Osborne 1990). As a result, a large majority of (~75%) *Q. engelmannii* seedlings occur under the canopy of mature con-specifics. The most of the remaining seedlings develop within 3m of the dripline (edge of the canopy) and are thus shaded for at least part of the day. Very few seedlings of *Q. engelmannii* survive in areas that receive direct solar radiation for the entire day (Griggs 1987, Lathrop & Osborne 1990).

In contrast to seedlings, most saplings of *Q. engelmannii* are found near the dripline or just outside the canopy of parent trees where sun light becomes less limiting. The dripline may also provide other benefits such as protection from herbivory and increased soil moisture due to canopy run-off (Lawson 1993, Lathrop & Osborne 1990, Muick 1991, Swiecki & Berhardt

1998). The transition from seedling to sapling is a rare event due to the high mortality of seedlings and the small number seedlings that recruit in areas conducive to sapling growth (Griggs 1987).

Direct effects of fire on Q. engelmannii

Q. engelmannii exists in an environment conducive to wildfires. The long hot summers with virtually no precipitation leads to large accumulations of dry fuels. Dry desert air in the form of Santa Ana winds blow west in the fall to further increase the probability and intensity of fire events.

Although *Q. engelmannii* exists in a habitat prone to wildfires, it is susceptible to heat and fire damage. Thin growing tissues on the surface of the trunks can be damaged even when the bark is not charred (Scott 1990). Mature Engelmann oak trees do not commonly resprout from the basal root crown. The inability of mature Engelmann oaks to sprout following hot brush fires may restrict its distribution to savannas and areas of open coastal sage scrub where high intensity crown fires are rare. Adult mortality due to understory fires is uncommon. However, trunk and crown damage readily occur even in cool grass-layer fires. In healthy individuals canopy sprouting occurs rapidly in areas receiving fire damage.

Unlike mature Engelmann oak trees, seedlings and saplings readily resprout from the basal root crown and belowground bud zones (Plumb 1980). Resprouting of immature oaks may be vigorous with growth rates almost twice the rate of unburned individuals (Mensing 1992). However, short fire return intervals may exceed the capacity of juvenile oaks to resprout and recover from repeated fires (Swiecki & Bernhardt 1998).

The survival of juvenile oaks following fire depends in part on pre-fire size. The effect of controlled burns on seedling survival under natural conditions appears detrimental. Second year survival rates for burned *Q. engelmannii* seedlings were 64% and 59% following two controlled burns. The survival rate for unburned seedlings was 80%. *Q. engelmannii* saplings experience much less mortality in response to these fires. The survival rate for burned saplings was virtually the same as that for control saplings, 95% and 97% respectively (Lawson 1993). High survival rates for *Q. engelmannii* saplings could be a result of their relatively thick bark, as reported by Snow (1972) or their ability to resprout vigorously after fire.

Indirect effects of fire on Q. engelmannii

Fire can also have indirect positive effects on oaks by impacting their local environment. In grasslands and savannas, spring fires reduce the density of non-native annual grass seed and potentially future grass densities. An increase in soil nutrients will result from the ash produced and deposited by the fire (Debano & Conrad 1978). Following fire events first year seedlings and resprouting individuals may experience reduced competition for soil moisture. Mature trees that are killed or sustain injuries will create gaps of reduced competition and increased light availability for seedlings and saplings. Reduced competition can enhance growth and increase the probability of oaks reaching maturity (Swiecki & Bernhardt 1998). A single fire event may be beneficial to seedlings and saplings that are able to resprout or escape major fire damage.

Santa Rosa Plateau

The Santa Rosa Plateau is located at the southeastern end of the Santa Ana Mountains in Riverside County, California. The reserve is dominated by mesas and rolling hills. The climate is typical Mediterranean with hot dry summers and cool wet winters. There are three main vegetation communities on the reserve: chaparral, grasslands and oak woodlands. The southern oak woodlands of the Santa Rosa Plateau (SRP) are composed of *Quercus agrifolia* and *Quercus engelmannii*. The Engelmann oak phase is primarily savanna, with a contiguous grass understory and scattered trees. This savanna landscape dominates on the dry mesa and hill tops.

Oak regeneration at the SRP has declined. Populations of Engelmann oak on the SRP appear to have insufficient recruitment to sustain current stand sizes. Past surveys of Engelmann oak stands have required large sample areas in order to locate even a few mature individuals with sufficient seedling and sapling numbers to carry out studies (Lathrop & Osborne 1990, Griggs 1987). Cattle grazing occurred from the 1840's (Snow 1972) until 1996 on some parts of the reserve. Three of the most common non-native species in SRP Engelmann oak savannas are *Bromus diandrus, B. mollis* and *Erodium botrys*. These three species are known to reduce growth and survivorship of oak seedlings (Gordon & Rice 1993). Areas of the SRP have high densities of ground squirrels and pocket gophers due to continued human disturbance and abundant *Erodium* (Snow 1972). These factors along with alterations to the natural fire regime have reduced Engelmann oak regeneration in oak savannas of the SRP.

Fire has been used as a management tool on the SRP under the management of The Nature Conservancy. The grasslands and oak savannas of the SRP are burned fairly regularly to maintain systems dominated by native perennial bunch grasses and native forbs. Spring burns are conducted to reduce the cover of non-native annual grass by reducing the input of new seed into the seedbank. The presence of fire, past grazing, highly competitive annual grasses and the dry climate has prevented the recruitment of woody species into the grasslands on a large scale.

The SRP offers an excellent opportunity to study the effects of fire on Engelmann oak regeneration due to the active fire management plan. The burn plan was established to return natural processes to the reserve and to restore and maintain native dominated grasslands and savanna understory. The effects of the current fire regime at the SRP and of fire in general on the population dynamics of *Q. engelmannii* are not well understood. Increasing the understanding of how fire affects Engelmann oak demography is critical in managing for sustainable populations. A series of observational and experimental studies tracking the performance of individual oak seedlings and saplings was designed to determine how the population responds to controlled burning.

Methods and Materials

Two management burns were conducted on the SRP in early July of 1998. The burns were preformed to reduce non-native annual grass density in grasslands and savannas. The initial burn was carried out in the Windmill (WM) burn unit and the second was in the Sylvan Meadows (SM) burn unit. Both burn units contained areas of oak savanna with sufficient numbers of *Q. engelmannii* seedlings and saplings to investigate the effects of fire on demography. Both fires burned under low to moderate intensities due to the relatively high fuel moisture levels. The relatively high moisture levels were due to the 1998-1999 El Niño event.

The two burn units vary in their recent land-use and fire histories. WM burn unit was previously burned in the spring of 1993. Additionally, WM and its control areas have not been grazed for about 15 years. SM burn unit and its surrounding control areas have not been exposed to fire for decades, however, these areas were grazed until the summer of 1996.

The effect of fire on seedling and sapling survivorship was investigated by comparing the response of burned individuals to individuals in unburned control areas. All saplings located were included in the study due to relatively low numbers. Saplings were defined as individuals greater than 50cm tall or with a stem diameter of 0.26cm or greater. The control sites were established in Engelmann oak savanna adjacent to the two burn units and matched by elevation, understory vegetation, slope, soils and aspect. For seedlings, five unburned control sites were selected for each burn unit. Each control site contained between 30 and 35 tagged seedlings.

Performance was evaluated by tagging individuals prior to the burns and conducting follow up surveys to check for mortality. At the time of tagging, height, number of live stems, the diameter of all stems, and canopy location were recorded. Immediately following the fires, burned sites were revisited to evaluate damage to seedlings and saplings. Damage was scored based on remaining aboveground structures: no damage (scored as a 4), leaves singed or blackened (3), leaves singed and blackened with at least 15% consumption (2), only the burnt stem remained (1), or no remaining aboveground structures (0). After seven months, sites were checked for additional mortality of control and undamaged individuals, resprouting of top-killed individuals, and subsequent mortality of resprouts.

Multiple logistic regression analysis (MLR) was used to assess the strength of the association between the independent variables and the probability of seedling and sapling survival. The factors investigated were site location (WM or SM), treatment (within control or burned area), pre-fire size, canopy location, and amount of fire damage. A stepwise MLR was used in model selection.

Effects of fire intensity

The effect of fire intensity on seedling survival was investigated by experimentally manipulating fuel loads around groups of seedlings. Aboveground herbaceous biomass was selected as the additional fuel to be added to the elevated fire intensity plots. For each elevated fire intensity plot, an equal area of similar biomass cover was harvested and added to treatment plots just prior to each burn. Elevated fire intensity plots were paired with control plots in which biomass was left unmanipulated. Plots were paired based on proximity and similarities in seedling size, herbaceous cover, and canopy location. Treatment plots were restricted to a maximum size of $2m^2$ and contained between 18 and 35 seedlings. Prior to the burns seedlings were tagged and measured as described above.

Following the burns, all sites were checked for completeness of burn and to assess damage to the seedlings. Damage to all seedlings was recorded based on remaining aboveground structures as in the previous portion of the study. Following the burns, sites were revisited at three month intervals. Multiple logistic regression analysis was used to assess the association between fire intensity and the probability of seedlings resprouting.

Effects of fire and herbivory

Quercus engelmannii acorns were planted in a factorial experiment with three levels of biomass removal (burned, mowed, and control) crossed with two levels of herbivory (protected and unprotected; Figure 1). Herbivore exclosures were constructed of a 1 cm diameter chickenwire cages 60cm tall buried to a depth of 30cm. A total of 450 pre-germinated acorns were planted in December of 1998; 150 at each of three sites. Five experimental units were planted per site, each containing 30 acorns from a single adult tree. Selection of sites was restricted to areas near the fire line to reduce distance between the burned and control portion of each site. Keeping the halves of the sites in relatively close proximity helped reduce differences in slope, aspect, elevation, and pre-fire vegetation. Sites were selected away from mature Engelmann oak trees to minimize variation in solar radiation reaching sites.

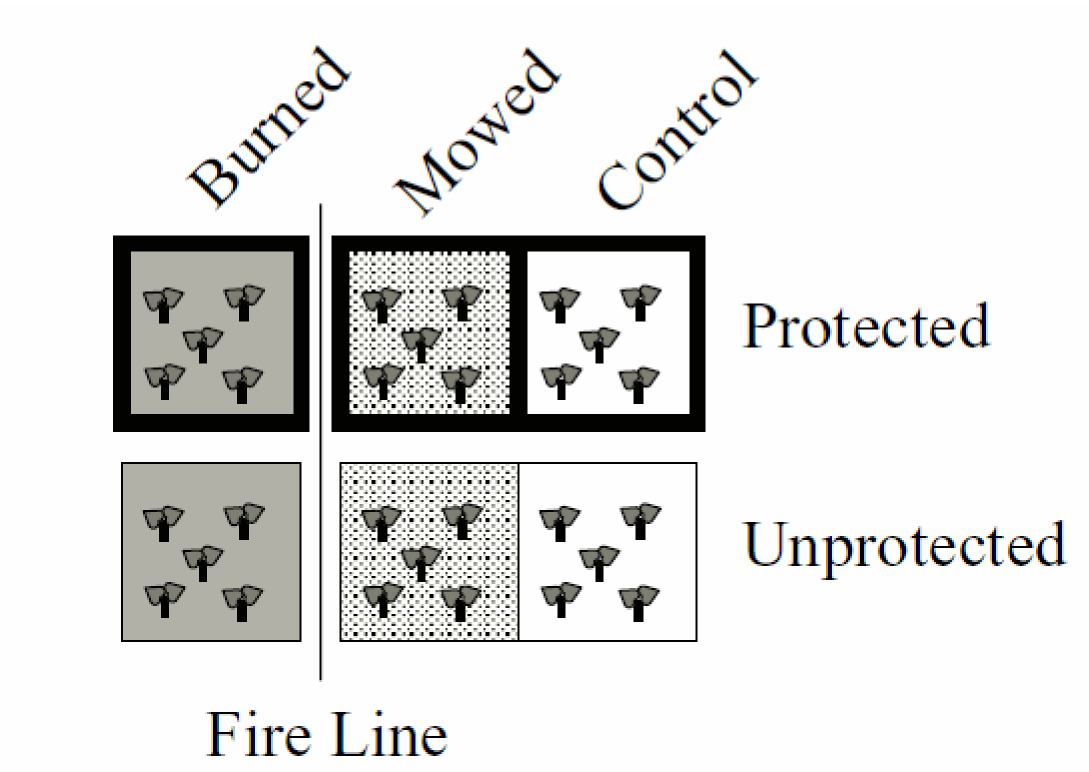

Figure 1. Diagram of an experimental unit from the acorn planting experiment. Repeated 5 times per site, once for each acorn source tree. A total of 3 sites at SM.

Results

All results are based on the February 1999 follow up survey, seven months after burning. A total of 2950 seedlings and saplings were initially tagged. For this survey 2419 individuals were relocated. Of those found, 92% were seedlings. Seedlings were very abundant on the reserve in the spring of 1998 due to the heavy rains associated with the El Niño event. Precipitation for the rainfall year ending June 30, 1998 was 63 inches, approximately 3 times higher than average (Snow 1972). Very few new seedlings were observed the following year. 1999 received only 10 inches of precipitation. Overall, saplings survived at a rate of 77% while seedling survival was only 24%. Seedlings and saplings in control areas survived at a higher rate than burned individuals, 56% compared to 22%.

The results of the logistic regression indicate that individual survival was dependent on fire damage, pre-fire size, site, and canopy location ($p < .001$ for all terms). The McFadden's Rho-squared (ρ^2) of the model was 0.308 indicating very high explanatory power of the model. The McFadden's Rho-squared is analogous to the R^2 from a general linear model. It is more difficult to interpret the ρ^2 but values above 0.3 are considered very good (Agresti 1990, Hosmer & Lemeshow 1989). No significant interactions were found among these factors. The lack of significant interactions suggests that these factors are additive and can be described one at a time.

Fire damage was by far the best single predictor of survivorship accounting for 64% of the total explained variance of the four-variable model. Survivorship increased as the amount of fire damage declined. Pre-fire size was the second best predictor of survival accounting for 22% of the explained variance. Larger individuals generally survived at a higher rate than smaller individuals regardless of treatment (Table 1). Interestingly, control individuals had a lower rate of survival (56%) than either undamaged (75%) or lightly damaged (61%) individuals from the burn units. The primary reason for this was that a majority of undamaged or lightly damaged individuals in the burn units were saplings, while a majority of control individuals were seedlings. Fire damage was thus correlated to pre-fire size with larger individuals escaping high levels of damage as a result of thicker bark and the leaves elevated above the level of the flames.

Table 1. Results of the seedling and sapling survivorship survey based on February 1999 data. Survivorship is broken down by site, fire damage and pre-fire size. Sample size is provided in parentheses.

Sylvan Meadows	Small Seedlings		Large Seedlings		Saplings		All Size Classes	
Control	24%	(129)	38%	(61)	NA	(0)	28%	(190)
Light Fire Damage	26%	(140)	43%	(81)	86%	(36)	40%	(257)
Heavy Fire Damage	6%	(549)	11%	(203)	100%	(3)	8%	(755)
All Individuals	12%	(818)	23%	(345)	90%	(39)	18%	(1202)

Windmill	Small Seedlings		Large Seedlings		Saplings		All Size Classes	
Control	61%	(92)	86%	(104)	100%	(41)	79%	(237)
Light Fire Damage	48%	(25)	61%	(79)	67%	(73)	62%	(177)
Heavy Fire Damage	10%	(347)	25%	(420)	58%	(36)	19%	(803)
All Individuals	22%	(464)	40%	(603)	74%	(150)	37%	(1217)

Combined Locations	Small Seedlings		Large Seedlings		Saplings		All Size Classes	
Control	39%	(221)	68%	(165)	100%	(41)	56%	(427)
Light Fire Damage	29%	(165)	52%	(160)	73%	(109)	48%	(434)
Heavy Fire Damage	7%	(896)	21%	(623)	62%	(39)	14%	(1558)
All Individuals	19%	(1282)	34%	(948)	77%	(189)	28%	(2419)

Differences between the two sites explained 9% of the variance of the model. The survival rate for seedlings and saplings at WM was 37% compared to 18% at SM. However, more saplings were found in both the burn unit and control areas of WM. Canopy location (CL) explained only 4% of the model variance. Individual that were under the canopy survived at about half the rate (inner half 19%, outer half 26%) of individuals outside the canopy (51%). These differences in survival are large, but much of the difference in survival rates between canopy locations is due to the size distribution of seedlings and saplings. Small seedlings are most common in the inner half of the canopy and have low survival. Saplings occur most frequently outside the canopy and have high survival.

Seedling Fire Intensity

A total of 16 paired plots were used to compare survival rates of seedlings exposed to ambient and elevated fire intensity. A total of 886 seedlings were analyzed, 453 in control plots and 433 in the elevated fire intensity plots. Higher fuel loads and thus greater fire intensity resulted in slightly increased fire damage to individuals in elevated fire intensity plots. Fire damage averaged 0.53 in elevated fire intensity plots and 1.02 in control plots (5 point scale; 0 = no remaining above ground structures and 1 = charred stem). Surprisingly, survival between treatments turned out to be virtually identical, 16% for controls and 15% for elevated intensity plots.

Burn damage and survival were both extremely variable among pairs of plots. The difference in burn damage between pairs was often larger than the differences between control and treatment plots. This suggests that spatial heterogeneity in fire intensity is a key aspect of the burns.

Acorn Planting

A total of 450 pre-germinated acorns were planted to assess the effects of herbivory and biomass manipulation on seedling emergence. As a result of the dry spring of 1999, emergence was uncommon and only 57 seedlings emerged. The ability of seedlings to emerge was significantly affected by treatment ($p = 0.025$). The cleared plots resulted in the highest levels of seedling recruitment accounting for more than 50% of all successful seedlings (Figure 2). Most of the positive effect of clearing came from a single site, site 1. Cleared plots at site 1 accounted for 47% of cleared plots with seedlings and contained 58% of the seedlings within all cleared plots. Within this site, however, there were 5 replicate mowed areas, each about 20m apart. At sites 2 & 3 both the clearing and fire treatments had the highest number of plots with seedlings but low sample sizes make interpretation difficult.

The source of the acorns did not significantly affect the ability of seedlings to emerge. Interestingly, two of the five trees from which acorns were collected occurred within the SM burn unit and received fire damage to the lower parts of their canopies. The three remaining trees were located in unburned control areas. Fire damage did significantly affect acorn weights ($p < 0.001$). Acorns from damaged trees weighed 14% less than acorns from control trees (1.4g versus 1.2 g). The effects of fire damage on acorn viability appeared to be low as most acorns that were collected germinated regardless of source tree (personal observation). The difference in mean acorn weight did not significantly effect seedling emergence even under the dry conditions of the spring of 1999. The observed difference of 0.2g did not appear to be biologically significant to subsequent recruitment.

The construction of herbivore exclosures had no effect on seedling emergence. Exclosures were unsuccessful in preventing pocket gophers from entering and digging up acorns and seedlings. Pocket gophers were responsible for all but two acts of vertebrate herbivory. Mice dug up two acorns in unprotected plots.

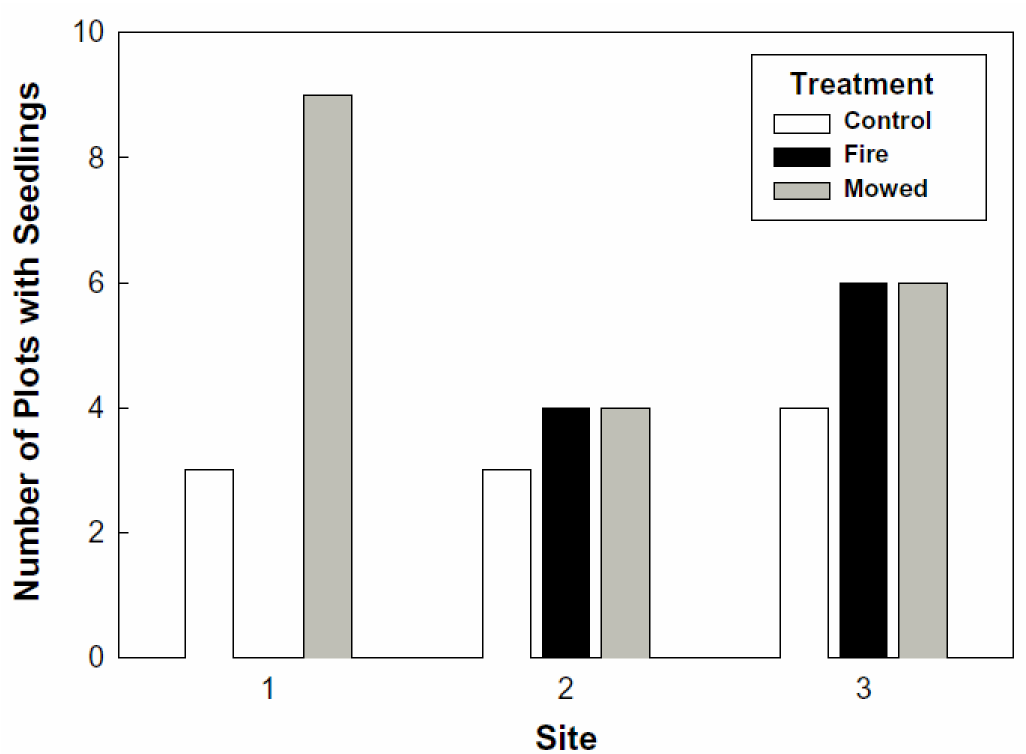

Figure 2. Distribution of plots with acorn emergence by site and treatment. The data from each herbivore exclosure plot and its paired unprotected plot were pooled due to ineffectiveness of exclosures and the small rate of seedling emergence. A total of 10 plots were planted per treatment per site, thus 90% of mowed plots at site 1 had seedlings. The number of plots with seedlings was significantly different between treatments, $p = 0.025$.

Discussion

Pre-fire size and the amount of fire damage incurred by Engelmann oak seedlings and saplings were the primary factors in determining survival. The response by Engelmann oak individuals to these factors indicates that burns should be conducted at times when fire intensity will be low and in areas containing larger seedlings and saplings. Restricting burns to areas containing larger seedlings and saplings should allow for adequate recovery by existing individuals following fire events.

Maintaining low fire intensity will be difficult due to the heterogeneity of fire behavior. Even burns carried out under conditions that minimize fire intensity will contain areas that burn hot due to variability in fuel, micro-climate and topography. In this study, increased fire damage associated with a doubling of dry biomass was small relative to the natural heterogeneity in fuel structure and distribution. High fuel loads, low fuel moisture and fuel structures that allow for rapid oxygen consumption will all locally increase fire intensity. Additionally, fire intensity will generally be greatest when fire burns up steep slopes due to the pre-drying of fuels.

Season of burn may also play an important role in maintaining Engelmann oak woodlands. Spring fires will generally burn cool and when carried out prior to seed drop by non-native annual grasses will reduce future competition with oaks. However, one study found that spring burns may favor the establishment of *Q. agrifolia*, the coast live oak, over Engelmann oak in southern oak woodlands. *Q. agrifolia* is an important species in southern oak woodlands but has a much larger distribution that *Q. engelmannii* and is generally reproducing at rates that will maintain current stand densities. As a result, fall burns conducted after the first rain may be desirable in areas with potential for high rates of coast live oak invasion into areas now dominated by Engelmann oaks (Lawson 1993).

In areas lacking current Engelmann oak recruitment, a series of spring burns may be most beneficial to future recruitment. Fire can be used under conditions that will reduce competitors of oak seedlings while not eliminating current recruitment potential (acorn crop). Cool fires in mid to late spring are optimum conditions for maximizing non-native annual grass control and minimizing fire damage to trees. In areas of oak woodland restoration, the ability of damaged trees to produce viable acorns may be important. Better rates of seedling emergence at cleared and burned sites also indicate that if acorns become buried they will recruit best in areas cleared of aboveground biomass and with lower cover of competitors.

Analysis of vegetation cover data showed that treatment significantly effected the amount of bareground in plots ($p < 0.001$). In this study, burned plots averaged 24% bareground, mowed plots averaged 12%, and control plots averaged only 4%. Cover of non-native annual grasses did not differ between mowed and control plots (41%), but was lower in burned plots (27%). The results show that spring burns can reduce the cover of non-native annual grasses, but that a reduction in these species does not necessarily correspond with increased oak recruitment. From the results of the vegetation analysis it appears that reduced levels of cover, regardless of species, correspond best with the highest levels of oak recruitment.

The differences in survival rates and the number of saplings at the two study sites, SM and WM could be a result of different recent land-use histories. The lower number of saplings and lower overall survival rate of seedlings and saplings at SM may be the result of recent cattle grazing. SM was grazed up until the summer of 1996, and had not burned in approximately 50 years. Grazing was stopped at WM in 1986 and two spring burns have been conducted in the last 14 years under the management of The Nature Conservancy. The SM oak woodlands had only 41 saplings in 250 acres. In contrast, WM contained 179 saplings in an area of only 100 acres. The differences in sapling densities at the two sites may be the result of recent grazing histories.

The better age class structure and higher survival rates of oaks at WM are most likely a result of a longer amount of time since grazing. The negative effects of grazing are well documented and have probably been responsible for preventing oak regeneration at SM (Mensing 1992, Borchert et al. 1989). A large majority of the saplings found at SM were associated with rock outcrops or dense shrubs. This may indicate that recruitment has been restricted to areas not accessible to cattle browsing and trampling. At WM recruitment is primarily found near the drip line of mature trees and does not correspond with rock outcrops

Conclusions

The use of fire as a management tool is complex. The timing and conditions under which controlled burns are carried out have a profound effect on the response of the target ecosystem. Goals of the fire management plan should therefor be defined before implementation. In Engelmann oak savannas, non-native annual grasses, native and non-native forbs respond negatively to spring burns. Both native and non-native forbs respond positively to fall burns. In addition, it has been found that Engelmann oaks respond better to fall burns, while coast live oak respond best to spring burns (Lawson 1993). A goal to maximize native species diversity or enhance Engelmann oak recruitment (in areas currently containing seedlings and saplings) would best be met with fall burns. A goal to reduce the cover of non-native annual grasses to reduce future competition with Engelmann oak seedlings (in areas currently lacking seedlings and saplings) would best be met with spring burns.

Fire behavior is highly unpredictable and no two fires are the same. The grazing and fire history of an area helps determine fire behavior by influencing fuel loads. Recent climate conditions also influence fuel loads by determining the amount of vegetation growth. Weather conditions at the time of the fire are critical in influencing fire behavior. Wind, humidity and temperature are important determinants of fire intensity, as well as, of the rate and direction of spread. When we discuss the use of fire as a management tool we must be careful not to forget that variations in the season and frequency of fire will greatly affect the outcome of this management activity.

In addition to variation in fire behavior due to the timing of the fire, great variation can also occur within a single fire event. A single fire event will be heterogeneous across the landscape resulting in a mosaic of differentially burned areas. For this reason it is important to understand the objectives of the fire and plan accordingly. Some areas will benefit from backing fires to reduce fire damage to critical resources. Other areas will need the extreme conditions of a head fire to burn up undesirable non-native species. In the management of Engelmann oak woodlands it appears that cool patchy fires may benefit oak regeneration by minimizing damage to existing individuals and allowing mature trees to complete acorn production. Spring burns maybe best suited for areas with dense stands of non-native annual grasses and low current recruitment. These burns will reduce future competition with developing oak seedlings and saplings, as well as, keeping fire intensity low to minimize fire damage to acorn producing trees. Fall burns may help maintain current stands and prevent the invasion of coast live oaks on a large scale. As with any woody species Engelmann oak seedlings and saplings will require time between fires to recover and grow to the next size class if current stands are to be maintained.

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Growth of Ponderosa Seedlings Following Preharvest Burning

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Abstract

The efficacy of fire as a tool to aid in regeneration and growth of tree stands is continually questioned. To address this problem, plots were set up to measure the response of naturally seeded ponderosa pines following burning and mechanically treating areas in a pine/ceanothus/sedge type. Growth of the natural regeneration on plots burned once and twice before harvest was compared with natural regeneration on a mechanically prepared site as a control. The unreplicated experiment was confounded by years as the control was cut as a shelterwood seven years before the preharvest burn plots were harvested. Nine years after harvest of the burn plots, growth of pines on plots of 1,200 square meters in each of the once- and twice-burned areas was significantly greater than the same sized mechanically-treated control for the past five years.

Introduction

The use of fire before harvesting has been proposed to reduce the competition with conifer seedlings from shrubby or herbaceous vegetation (Biswell 1982 , Martin 1982, Kauffman 1986). To date, there has been no study carried through from preharvest burns to the growth of seedlings after harvest. The purpose of this paper is to report on the results of an unreplicated preharvest burn trial. The trial was the precursor of a much-replicated study that was dropped when the senior author moved. Since there is no other report available to indicate the usefulness of the technique, this unreplicated study assumes importance it otherwise would not have had.

Methods

The study was conducted on the Lookout Mountain Unit of the Pringle Falls Experimental Forest in central Oregon. The elevation of the study reported here is about 1,495 meters (4,900 feet) on a Typic Cryorthant pumice soil. The site is classified as mixed conifer-ceanothus-sedge, USDA Forest Service type CW-S1-15 (Volland 1976). Annual precipitation averages 63 to 89 cm (25 to 35 inches), mostly as snow during the winter. Site index for ponderosa is about 29 meters (95 feet) on a hundred year basis using Meyer's (1938, rev.1961) data for natural stands.

The site was occupied by an even-aged ponderosa pine (*Pinus ponderosa*) stand which originated about 1840, probably after a fire. Another fire had occurred on the site when the stand was about 40 years old, and might have thinned the stand. Occasional lodgepole pine (*Pinus contorta*) occurred in openings in the stand, and white fir (*Abies concolor*) were beginning to establish themselves in the understory. Shrubs consisted mostly of *Ceanothus velutinus* and *Arctostaphylos patula*, with occasional patches of chinkapin (*Castanopsis chrysophylla*). Procumbent shrubs included squawcarpet (*Ceanothus prostratus*) and princespine (*Chimaphila *umbellata*). The only common herbaceous plant before harvest was long-stolon sedge (*Carex pensylvanica*).

Units in the ponderosa pine stand were first burned in 1976 as part of a nutrient cycling study (Nissely 1978, Nissely et al 1980). The burn units originally consisted of three replications of burns conducted to 1) consume the foliage of the shrubs, or 2) to girdle the shrubs but not consume the foliage. On the plots reported here, an average of 19 and 31 tonnes/acre of litter, duff (Oe and Oa layers) and understory were consumed by the first burn on the once and twice burned plots, respectively. A second burn was conducted in the late summer of 1979 on half of two of the blocks. The second burn was designed to consume as much of the duff (Oe and Oa layers) as possible (Martin 1982). Following the second burn, over 90% of the old ceanothus shrubs had died without resprouting (Martin 1982). Because of the downturn in the timber market, only one block was harvested, and that because of its proximity to the shelterwood used as the control in this study. The scattered slash on the preharvest burn plots was broadcast burned in the summer of 1981.

The control unit (no pre-harvest burn) was cut as a shelterwood in 1974 with about 20 shelter trees per acre. The slash was mechanically piled, and then burned in the fall of 1975. Shrubs were mechanically removed with a brush rake and piled with the slash, giving the site a very thorough site preparation treatment. It was with the removal of the shelterwood in the winter of 1980-1981 that the preharvest burn units adjacent to the shelterwood were logged.

The regeneration and competing vegetation consisted of 30 by 40 meter or 0.12 hectare (98.4 by 131.2 feet or 0.296 acres) plots. The size was determined by the limits of the preharvest burn units, leaving a 10 meter buffer from the roads and adjoining older stand. One hundred trees were selected in each unit, the selection being based on "the trees I would leave in a first precommercial thinning in about five or ten years." This would leave an average of about 12 square meters (129 square feet) per tree, or a spacing of about 3.46 meters (11.4 feet). The 100 trees selected did not mean the 100 best trees were selected. Many good trees were not included because of their proximity to selected trees which were larger or better distributed.

For each tree selected, the total height and diameter at 11.4 cm (4.5 inches) above the ground were measured. The length of each internode, starting at the top, was measured for as many internodes as could be determined. Competing vegetation within one meter of each measured tree was estimated as to the fraction of the one meter radius the species of plant occupied and its average height.

Results and Discussion

Growth of Pines and Competition

A plot of the data indicate that for the last five years, 1986 to 1990, the two preharvest burn treatments outgrew the mechanically prepared (control) treatment (Figure 1). Analysis of variance indicated the differences were very highly significant ($P<0.001$, Table 1). For the years 1990 back through 1986, the preharvest burn plots outgrew the control by highly significant amounts. For the years 1985 through 1983, the reverse was true. No statistical comparisons were made for the years previous to 1983, as data for the two burn units were too sparse. They had been harvested only in the winter of 1980-1981, and little regeneration was established.

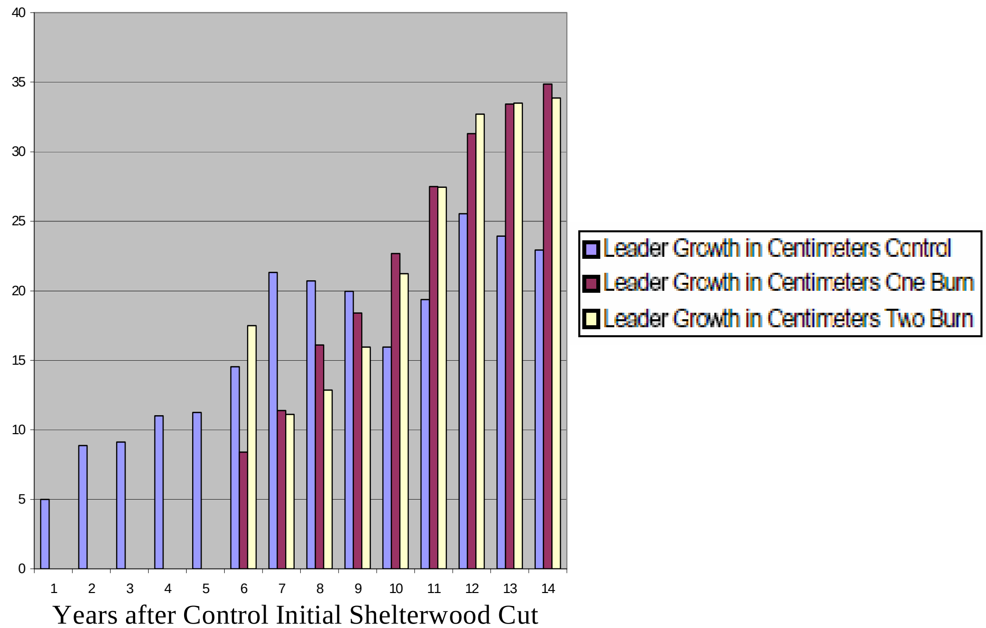

Figure 1. The one (1 PHB) and two (2 PHB) plots had greater growth for the last five years than did the no burn (NO PHB) control plot. For the three years before that, the reverse was true.

The slower growth in the first few years of a seedling's life would be expected on most sites. On this site, one would expect growth to show a larger increase in rate after a few years once the roots reach and utilize the buried soil profile overlain by the pumice.

The results are confounded by the earlier shelterwood harvest of the control unit. The regeneration on it was established very soon after the 1974 harvest. At least some of the regeneration would have been hindered by competition with the shelter trees until their removal in the 1980-1981 winter, when the burn units were also harvested.

Observations over the years indicated the control was very clean of competing shrubs for the first two to three years after the original harvest. After the initial period, however, the shrubs, particularly greenleaf manzanita, seeded in very rapidly. The manzanita often overtopped the tree seedlings.

In the burn units, the shrub regeneration seemed to be delayed for a longer time, and the composition of the shrubs seemed to contain more ceanothus. Large patches of long-stolon sedge are also present in the burn units, while not so prevalent in the control. This latter difference could stem from original differences in composition or from the sedge being rooted out by the mechanical preparation in the control unit. The sedge seems to hinder growth of the tree seedlings, but they will survive and continue to grow once established in the sedge patches.

The ceanothus velutinus began to grow a few years after the pines regenerated on the burn units. It is particularly prominent in a ten meter wide strip bordering the uncut pine stand. Where the ceanothus has overtopped the pine, the pine continues to grow well, but not as fast as the pine without heavy competition. It appears the ceanothus is not that serious a competitor once the pine is well-established, and the pine should outgrow the ceanothus in a few years. Perhaps the young ceanothus is also increasing the nitrogen available to the pines (Youngberg et al 1979).

Seedling Density

The pine seedlings on the mechanically prepared control are much denser than on the preharvest burn units, averaging probably less than 60 centimeters (2 feet) in spacing. The spacing of seedlings on the preharvest burn units is estimated at an average 1.5 to two meters. Thus, inter-tree competition will become a problem earlier in the control unit and require heavier and more expensive thinning than in the preburn units. The overstory, judging by the size of the branch stubs, had grown as a widely spaced stand. The wider spacing of the overstory stand might have resulted from the fire which occurred in it at about age 40.

Conclusions

The one and two preharvest burn treatments seem to provide a better environment for the growth of natural pine regeneration than does the mechanical site preparation, although both resulted in adequate stocking. The confounding of treatments and years makes comparison difficult.

Stocking on the mechanically prepared site was very dense, which will require a very early precommercial thinning to relieve the overstocking and resulting stagnation of the stand.

Table 1. - Annual leader growth and significant difference in growth of ponderosa pine following harvest for control, one and two preharvest burns. Identical letters indicate no significant differences for any given year.

YEAR	CONTROL Growth - cm	ONE PREHARVEST BURN Growth - cm	TWO PREHARVEST BURNS Growth – cm
1990	22.93[a]	34.87[b]	35.86[b]
1989	23.94[a]	33.43[b]	33.50[b]
1988	25.54[a]	31.31[b]	32.70[b]
1987	19.35[a]	27.50[b]	27.46[b]
1986	15.95[a]	22.68[b]	21.23[b]
1985	19.97[a]	18.40[a]	15.96[b]
1984	20.70[a]	16.09[b]	12.84[c]
1983	21.32[a]	11.38[b]	11.10[b]
1982	14.54	8.40	17.50
1981	11.24		
1970	11.0		
1979	9.12		
1978	8.87		
1977	5.0		

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A Meta-Analytic Model to Predict Fire Effects on Dominant Vegetation

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Abstract

Historic fire regimes are widely advocated as guides for the reintroduction of natural fire processes into North American wildlands. However, the structural effects of prescribed fire activities are largely unpredictable. Wide variability in fire effects, limits the generality of information provided by a single study or a narrative review. Fortunately, some generalization of fire effects is possible through meta-analysis: a quantitative synthesis of research results. I introduce a meta-analytic model that predicts fire effects on the abundance of the dominant plant species at a site. Required inputs include pre-fire mean abundance (biomasss or percent cover) of the dominant species at the site, historic ecosystem fire regime, and recovery time. The model produces a confidence interval for the post-fire mean abundance of the pre-fire dominant species, under the assumption that the weather remains constant.

The predictors included in the model explain 62 percent of the random variation in outcomes from fire effects studies that were controlled for climatic variations. Fire behavior was excluded as a predictor due to inconsistent reporting in the literature. The effect of fire type (prescribed versus wild) appears significant in some ecosystems, but it was not included as a predictor because its effect was generally not assessable (e.g., all studies conducted in low frequency/high intensity ecosystem fire regimes were of wildfires). The effect of burn season was also insignificant and excluded from the final model.

The significance of the meta-analytic fire effects model provides statistical confirmation of the need to consider fire history in planning fire management activities. Though the model is very general and limited to a single species, it may provide managers a useful indication of what to expect from future fires.

Introduction

Researchers have published thousands of fire effects studies (Fischer et al., 1996) that provide information essential to sound fire management. Knowledge of fire effects on natural resources is necessary for wildfire damage assessments (Crosby, 1977) and successful prescribed burns (Zimmerman, 1994). Literature reviews (e.g., Lotan et al., 1981), fire ecology books (e.g., Agee, 1993) and the Fire Effects Information System accessible over the Internet (Fischer et al., 1996) highlight the need for syntheses of the fire effects literature.

However, all extant fire effects research syntheses rely on qualitative comparisons to reach general conclusions. The power of this approach is limited, especially when research results depend on factors that vary between studies (Gurevitch et al., 1992). Separation of real fire effects from other variables is a problem widely acknowledged in narrative reviews (e.g., Whelan, 1995, p.4).

Meta-analysis (MA) offers an alternative approach to research synthesis: statistical procedures for analyzing results from primary research. Meta-analysis provides an empirical assessment of the consistency and significance of results reported in different studies (Cooper and Hedges, 1994). Classification of studies by characteristics believed to influence observed effects allows comparison of within-class variation to between-class variation with procedures that are analogous to ANOVA. Though the quality of studies upon which an MA is based will determine the resultant degree of clarity, MA may provide the best means for separation of real effects from random error in fields where replication is difficult or impossible (Gurevitch et al., 1992), such as fire ecology.

The use of MA is widespread in medicine and psychology, but ecologists have just begun to explore its applications (Brett, 1997). Gurevitch et al. published one of the first ecological MA's in 1992 with a focus on the effects of competition. More recently, the journal Ecology devoted much of its June 1999 issue to ecological MA. Here I introduce MA to fire effects research with a focus on changes to biomass or area coverage of dominant plant species. My primary objective was to quantify the heterogeneity in reported fire effects and determine the amount of variation explainable by differences between studies in ecosystem type, recovery time, type of fire, and season of burn. The MA resulted in a general model that may provide managers a useful indication of what to expect from fire events.

Methods

An MA comprises four basic procedures: literature search for studies to include in the analysis, classification of studies by influential characteristics, calculation of a standardized effect size for each study, and assessment of the significance of differences between classes of studies (Cooper and Hedges, 1994).

Literature Search

The literature search relied upon the 1970-1996 AGRICOLA (Cambridge Scientific Abstracts, 1996) database for identification of fire effects studies on plants. The MA includes all citations found by this method that met the following selection criteria: (*i*) Conducted in North America. (*ii*) Reported in English. (*iii*) Published in a peer-reviewed journal, conference proceedings, or as a scientific report to or by a federal agency. (*iv*) Results reported separately for individual species. (*v*) Responses measured as coverage or biomass. (*vi*) Included contemporaneous measurements from an 'unburned' reference area to control for climatic influences on post-fire responses.

Within the bounds of these literature search and selection criteria the MA is comprehensive of all fire effects studies. However, it is likely that the search method overlooked some studies, that relevant unpublished studies exist, and that there are unstudied fire effects. Therefore, the MA probably is not representative of all possible fire effects on dominant plant species abundance.

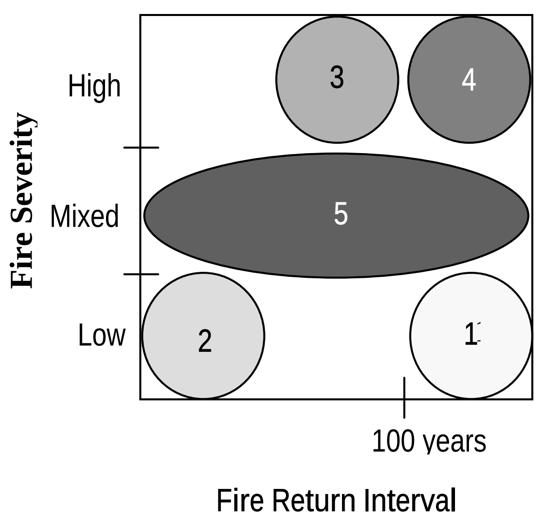
Study Classification

The studies included in the MA were classified by each of four categorical variables: ecosystem fire regime, recovery time, fire type, and season of burn. Explicit criteria for each variable ensured consistent and unbiased classification.

Ecosystems have evolved under a variety of fire types and frequencies such that species may exhibit adaptations to specific fire regimes (Agee, 1993). The ecosystem in which each study took place was classified into one of the five general fire regimes described in Figure 1. This fire regime classification is essentially Sando's (1978) with the addition of a mixed regime. The fire regime classification used in this MA is an artificial construct that provides a convenient grouping of ecosystems for comparative purposes. Assignment of ecosystems to fire regime classes was based upon relevant fire history studies referred to by the study author or in one of the fire regime discussions by Heinselman, Kilgore, Kucera, or Christensen contained in Mooney et al. (1981). Classification of ecosystems with unknown fire history was based on the fire regime of ecosystems with similar geography and vegetation. Ecosystems with no applicable fire history information were assigned to a sixth class that was not included in assessments of the influence of ecosystem fire regimes on fire effects.

Fire effects studies may take place any number of years after a fire occurrence. Recovery time was classified as one of three periods: short (<11 years), intermediate (11-50 years), or long (51 to 100 years). Areas that have never experienced fire are probably very rare and it is likely that all fire effects studies compare two burned areas of different ages, with the older considered an 'unburned' control. An older burn area was considered a valid control for the assessment of fire effects on a younger burn only if the two areas were in different time period categories.

Figure 1. Classification of ecosystem fire regimes. The numbers in the schematic below correspond to the fire regime classes in the table to the right. Studies were classified into one of five fire regimes based on the fire severity and frequency characteristic to the ecosystems in which they took place. Ecosystems with no applicable fire history information were assigned to a sixth class, which was not included in assessment of the influence of ecosystem fire regimes on fire effects.



Fire Regime Class	Ecosystems included in the meta-analysis
1	Deserts in CA and AZ Arctic tundra in the Northwest Territories Boreal lichen woodland in Quebec
2	Prairie in IA, OK, TX Pine savannahs in FL, GA, SD Oak savannahs in IL, MN, TX Southern pine forests in AL, AR Ponderosa pine forests in AZ, CO, ID Aspen parkland in Alberta
3	Pinyon-juniper woodland in UT Sagebrush in UT and MT Coastal scrub in CA and FL Coastal heath in NJ Chaparral in CA Birch forest in Labrador
4	Boreal mixed forests in the Northwest Territories, Labrador, and Quebec Subalpine white spruce forest in Alberta White fir forest in CA
5	Douglas-fir forest in CA Red pine forest in VT
6	Oak scrub in PA *(regime undetermined)*

Fire behavior should also influence species responses to fire events. However, descriptors of fire behavior are inconsistently reported in the fire effects literature and attempts to classify studies by this variable proved unsuccessful (see Martinson, 1998). Instead, studies were classified by fire type: prescribed or wild. Though fire type is an inadequate surrogate for fire behavior, there may be gross differences between the effects of prescribed and wild fires that are of interest from a management perspective.

The season in which a fire occurs may also influence fire effects due to seasonal differences in soil, fuel, and foliar moisture, fire behavior, and plant phenology (Agee, 1993). Season of burn was classified as spring (March to June), summer (June to September), autumn (September to December), or winter (December to March).

These four classification variables (ecosystem fire regime, recovery time, fire type, and season of burn) are considered the predictors of fire effects within the context of this MA. Differences between studies not explained by these classification variables were incorporated by inclusion of a random variance component (see below). The fire effect response variable is the biomass or area coverage of the dominant species observed in a particular study. Biomass and area coverage are allometrically related measures and the most meaningful indicators of competitive success when comparing species with various growth forms (Brower et al., 1990). The dominant species observed in a study was that which had the greatest biomass or area coverage in the study's unburned area. The dominant species was deemed the single best representative of community response to fire: a single species represented each study to avoid excessive non-independence in the MA, as well as unjustified weighting of studies that reported measurements on greater numbers of species.

Effect Size Estimation

Meta-analysis depends on an estimate of effect size (ES): the magnitude of the experimental treatment mean (the mean biomass or coverage measured in the burned area) relative to the control treatment mean (Cooper and Hedges, 1994). Calculation of ES's standardizes studies and allows their comparison, but normally requires that each study report a mean with a measure of variance: information commonly omitted in fire effects publications. Recently, however, Hedges et al. (1999) developed the log response ratio as a measure of ES that does not necessarily require studies to provide variance information. The log response ratio $ln(R)$ for a given study is calculated as

(1) $$ln(R) = \ln(X_b / X_u) \ ,$$

where ln is the natural log, X_b is the mean biomass or coverage in the burned area of the dominant species in the unburned area, and X_u is the mean biomass or coverage in the unburned area of the dominant species in the unburned area. Means reported in each study were extracted from tables or manually digitized from graphs with an engineer's scale.

Statistical Analysis

The paucity of variance information available in the fire effects literature precluded the use of standard procedures in this MA. However, the development of non-parametric resampling procedures for MA by Adams et al. (1997) did allow a less conventional analysis. Unweighted class means and 95% bias-corrected bootstrapped confidence intervals were calculated under mixed-effects model assumptions with MetaWin software (Rosenberg et al., 1997). MetaWin was also used to test for heterogeneity in fire effects between different classes of studies by randomization with 4,999 iterations.

This MA did not follow the convention of weighting studies by their inverse variances due to the general unavailability of the variance information that allows calculation of optimal weights. The MA did employ a mixed-effects model, which is considered most appropriate for MA's of ecological data (Gurevitch and Hedges, 1993). Mixed-effects models incorporate a random variance component that accounts for random effects due to inherent differences between studies grouped together by the fixed effect classification variables (i.e., ecosystem fire regime, recovery time, fire type, and season of burn). Comparison of the size of the random variance component (σ_θ^2) when study classification variables are included in the model to its size when they are left out provides a measure of the explanatory power (R^2) of the mixed effects model (Cooper and Hedges, 1994):

$$\textbf{(2)} \qquad \boldsymbol{R^2} = \frac{[\sigma_\theta^2 \text{ (no classification)} - \sigma_\theta^2 \text{ (classification variables included)}]}{\sigma_\theta^2 \text{ (no classification)}}.$$

See Rosenberg et al. (1997) and Cooper and Hedges (1994) for further statistical explanation of meta-analytical procedures.

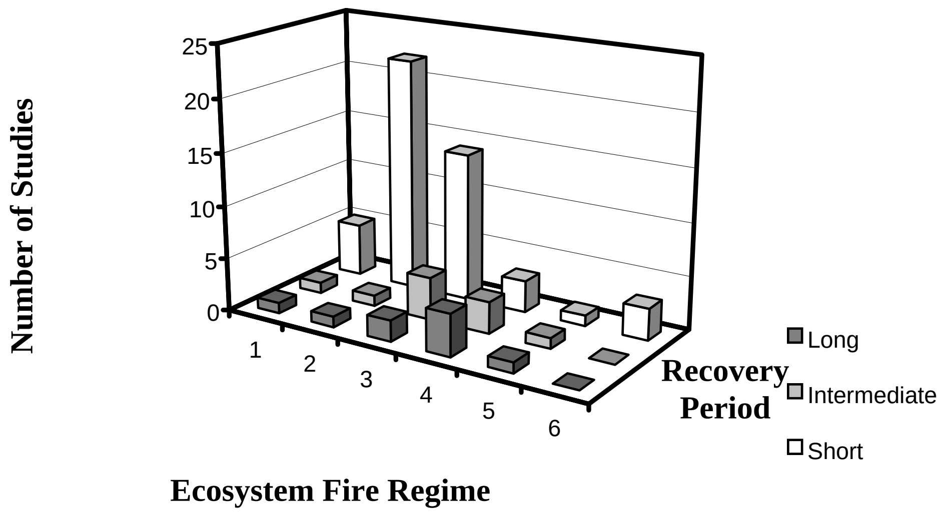
Results and Discussion

Table A1 in the Appendix lists the Forty-five publications identified in the literature search that met the selection criteria for the MA. Several articles provided comparisons for multiple fires, recovery periods, or fire regime classes, such that a total of 67 comparisons were included in the MA.

Study Characteristics

Studies were unevenly distributed across the ecosystem, recovery time, fire type, and burn season classification variables. Most studies reported results in ecosystems with intermediate to high frequency characteristic fire regimes (Regimes 2 and 3: 66%) and after short recovery periods (72%). The number of prescribed fire studies and wild fire studies were nearly equal. However, prescribed fire studies were concentrated in high frequency, low severity fire regime ecosystems (Regime 2: 64%) and after a short recovery period (97%). Only prescribed fire studies consistently reported information on season of burn and most occurred in spring (44%). The overall distribution of the 67 comparisons across the ecosystem and recovery time classification variables is provided in Figure 2.

Figure 2: Distribution of studies across the Fire Regime and Recovery Period classification factors. See Figure 1 for explanation of the six Fire



Heterogeneity of Fire Effects

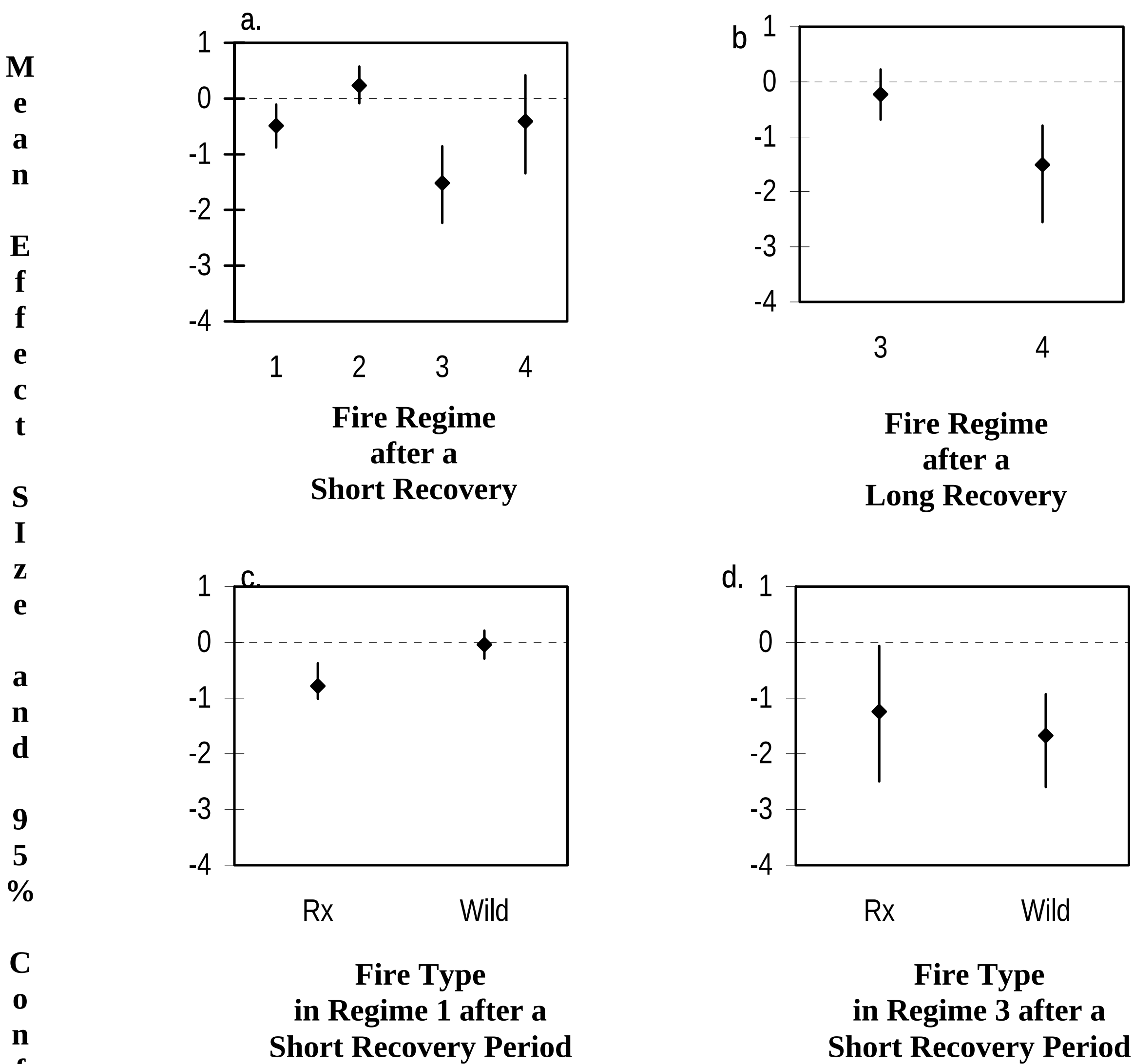
Fire causes a significant decrease in the biomass and coverage of dominant species when averaged across all ecosystems, recovery periods, fire types, and seasons of burn. The abundance of dominant species in burned areas is half that in comparable unburned areas, on average, with an ES of *lnr* equal to –0.67 and a 95% confidence interval (CI) that does not include zero (-1.02< *lnr* < -0.36). However, observed fire effects are highly heterogeneous between studies (p<0.001). Much of this heterogeneity is explained by the study classification variables.

Significant differences in the response of dominant species to fire were found in comparisons between ecosystem fire regimes after both short and long recovery periods. Mean abundance of dominant species decreases after a short recovery period from fire, except in the low severity, high frequency Regime 2 (Figure 3a). This result is not unexpected, since species in this type of ecosystem are most likely to exhibit adaptations for fire resistance (Agee, 1993).

After a long recovery period only Regimes 3 and 4 included enough studies to make comparisons. Figure 3b shows that fire effects on dominant species become insignificant after a long recovery period in the higher frequency Regime 3. Dominant species in the lower frequency Regime 4, in contrast, may take more than a century to recover from fire events. It is evident from Figures 3a and 3b that recovery period interacts with ecosystem fire regime to influence fire effects (p=0.008).

Fair comparisons between the effects of prescribed and wild fires could be made only in ecosystem fire regimes 1 (low severity and frequency) and 3 (high severity, intermediate frequency) after a short recovery period. Other regimes and recovery periods contained only studies of one fire type or the other. Figures 3c and 3d show that prescribed fires and wild fires may have significantly different short-term effects on the abundance of dominant species in some types of ecosystems. However, the lack of information on the effect of fire type in most ecosystems marginalized its contribution to the explanatory power of the final model. Season of burn was also excluded from the final model, since differences between seasons of burn were not detected (p=0.75; Figure 3e).

Ecosystem Fire Regime



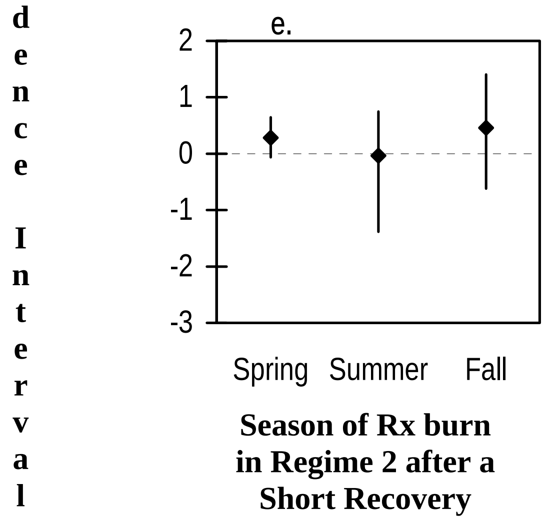


Figure 3. Mean fire Effect Sizes for several combinations of ecosystem fire regime (see Figure 1), recovery period, fire type, and season of burn. Mean Effect Sizes with confidence intervals that do not include zero are significant at the 0.05 level. Non-overlapping confidence intervals indicate significant differences between class means.

Meta-analytic Model

The interaction of ecosystem fire regime and recovery period explained the most variability in fire effects on dominant species and was chosen as the best predictor for the final model (R^2 = 0.62). Table 2 provides a 95% confidence range for the amount by which the abundance of dominant species would be expected to multiply as a result of fire for each ecosystem fire regime and recovery period. Note that these multiplication factors are relative to future abundance without fire: actual future abundance of dominant species after fire will depend upon weather conditions.

Table 2. Multipliers to predict dominant species abundance with 95% confidence, relative to expected abundance without fire.

		Recovery Period		
		Short	Medium	Long
Ecosystem Fire Regime	1	0.42 ↔ 0.90	0.01	0.17
	2	0.92 ↔ 1.78	0.33	0.55
	3	0.11 ↔ 0.42	0.04 ↔ 1.10	0.50 ↔ 1.10
	4	0.26 ↔ 1.52	0.08 ↔ 2.45	0.08 ↔ 0.45
	5	0.10	0.09	1.34

Note: Ranges in the table correspond to a 95% Confidence Interval. Cells with a single value indicate ecosystem and recovery combinations with only one applicable study.

Caveats

There are several characteristics of the fire effects literature which compromise the results and conclusions of this MA. Fire effects are generally inferred from post-facto comparisons between a burned area and a 'similar' unburned area. Since most fire effects studies lack random assignment of fire 'treatments' to sample units, pseudoreplication (sensu Hurlbert, 1984) is committed in any statistical tests. Fire ecologists are beginning to acknowledge the problems associated with pseudoreplication (Van Mantgem, this volume) and it will always present a barrier to strong statistical inferences about fire effects. An advantage of MA is that a degree of replication is provided through combination of ES's across multiple distinct studies. However, MA is intended for synthesis of experimental rather than observational studies and undoubtedly produces clearer results when based on ES's calculated from replicated data.

Another problem associated with inferences of fire effects from comparisons between burned and unburned areas is that neither area represents a control (i.e., an undisturbed state) in most ecosystems, particularly those that historically experienced frequent fire recurrence (Laudenslayer and Zack, this volume). Since an 'effect' cannot be assessed without a control, the meaning of comparisons between burned and unburned areas is ambiguous. However, the availability of alternative comparisons seems unlikely.

The general failure of fire effects studies to describe and account for variations in fire behavior most likely resulted in inappropriate study groupings and increased unexplainable variation in this meta-analysis. Our knowledge of fire effects will remain superficial as long as fire is treated dichotomously.

The omission of information about sampling variances from most fire effects studies necessitated a suboptimal meta-analysis based on unweighted resampling procedures (Gurevitch and Hedges, 1999). However, the methods used for this meta-analysis were probably most appropriate in any case, given the unreplicated nature of fire effects studies (Gurevitch, 1999)

Conclusions

Despite the aforementioned caveats, I conclude that MA can provide an effective quantitative synthesis of the extant fire effects database. This MA focused on the fire effects literature on plant biomass and coverage, but it demonstrates a useful approach for the MA of other fire research topics, such as 20th Century alterations to fire frequency and the effectiveness of fuel treatments (Martinson and Omi, 1999). The results of this MA support the common fire management supposition that fire effects vary between ecosystems with different characteristic fire regimes. This MA found that fire effects in high frequency fire regime ecosystems tend to be insignificant or short-term, while fire in low frequency fire regime ecosystems causes less ephemeral reductions in dominant species abundance. Finally, the MA resulted in a model that, though very general, provides some quantification of the effects that can be expected from future fire events.

Acknowledgments

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Appendix

Table A1. The 45 publications included in the meta-analysis of fire effects on plant abundance.

Author	Publication
Anderson and Bailey, 1979	Can. J.Bot. 57: 2819-2823
Barney and Frischknecht, 1974	J. Range Manage. 27: 91-96
Bentz and Woodard, 1988	Wildl. Soc. Bull. 16: 186-193
Bock and Bock, 1983	J. Wildl. Manage. 47: 836-840
Brown and Minnich, 1986	Am. Mid. Nat. 116: 411-422
Carlson et al., 1993	J. Wildl. Manage. 57: 914-928
Collins, 1987	Ecology 68: 1243-1250
Conard and Radosevich, 1982	Madrono 29: 42-56
Cox et al., 1990	USDA For. Serv. RM-GTR-191: 43-49
De Grandpré et al., 1993	J. Veg. Sci. 4: 803-810
Dhillon and Anderson, 1993	New Phytologist 123: 77-91
Engstrom and Mann, 1991	Can. J. For. Res. 21: 882-889
Foster, 1985	J. Ecol. 73: 517-534
Foster and King, 1986	J. Ecol. 74: 465-483
Fraas et al., 1992	USDA For. Serv. INT-GTR-289: 212-216
Gates and Tanner, 1988	Florida Scientist 51: 129-139
Greenberg et al., 1995	Am. Mid. Nat. 133: 149-163
Hadley and Veblen, 1993	Can. J. For. Res. 23: 479-491
Hallisey and Wood, 1976	J. Wildl. Manage. 40: 507-516
Hill and Platt, 1975	*Prairie: A Multiple View*. Edited by M.K. Wali. Grand Forks: University of North Dakota Press; 103-113.
Johnson and Abrahamson, 1990	Florida Scientist 53: 138-143
Johnson et al., 1986	Am. Mid. Nat. 116: 423-428
Keeley, 1992	J. Veg. Sci. 3: 79-90
Keeley and Zedler, 1978	Am. Mid. Nat. 99: 142-161
Landhäuser and Wein, 1993	J. Ecol. 81: 665-672.
Lewis and Hart, 1972	J. Range Manage. 25: 209-213
Malanson, 1984	Vegetatio 57: 121-128
Matlack et al., 1993	Biological Conservation 63: 1-8
Merrill, 1982	J. Wildl. Manage. 46: 496-502
Morneau and Payette, 1989	Can. J. Bot. 67: 2770-2782
Pase and Lindenmuth, 1971	J. For. 69: 800-805
Riggan et al., 1988	Ecol. Monogr. 58: 155-176
Schlesinger and Gill, 1980	Ecology 61: 781-789
Schmalzer and Hinkle, 1992	Castanea 57: 158-173
Soutiere and Bolen, 1976	J. Range Manage. 29: 226-231
Speake, 1966	Proceedings: Southeastern Game and Fish Commissioners 20: 19-32
Stuart et al., 1993	For. Sci. 39: 561-572
Svedarsky and Buckley, 1975	*Prairie: A Multiple View*. Edited by M.K. Wali. Grand Forks: University of North Dakota Press; 115-121.
Vogl and Schorr, 1972	Ecology 53: 1179-1188
Vose and White, 1987	Can. J. Bot. 65: 2280-2290
West and Hassan, 1985	J. Range Manage. 38: 131-134
White, 1986	USDA For. Serv. NC-RP-266: 1-12
Wilson et al., 1995	J. Wildl. Manage. 59: 56-67
Wink and Wright, 1973	J. Range Manage. 26: 326-329
Wright, 1974	J. Range Manage. 27: 417-419

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Brett, M.T. 1997. Meta-analysis in ecology. Bull. Ecol. Soc. Am. 78: 92-94.

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Microsite Recovery of Vegetation in a Pinyon-Juniper Woodland

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Abstract

An assessment of the short-term effects of prescribed fire on the understory vegetation of a pinyon-juniper woodland was conducted, with a focus on microsite recovery patterns with respect to dominant pinyon trees (*Pinus monophylla*). Data was collected for cover and density of all plant species during June of 1998, one year after treatment. Analysis of variance conducted on burn effects showed a significant decrease in cover for most perennial grasses, but no significant change of annual species. Non-graminoid response was variable. Most species showed no change or a significant decrease in density. Three disturbance/fire obligate species, absent from control plots, showed significant increases. Microsite effects showed species richness response was significantly lower under burned trees; recovery of perennial species in this region was only from species with well-insulated perennating tissues. We assume that, due to heavy fuel loading and more extreme fire effects under trees, non-adaptive perennials and seed were destroyed. Response of annuals in burns was essentially limited to inter-space areas. Due to elimination of competitive growth and a flush of nutrients following burning, total cover of cheatgrass (*Bromus tectorum*) increased 230% in the tree interspace region.

Introduction

One of the dominant plant communities within the intermountain region of the United States is the pinyon-juniper woodland, covering at least 7.1 million ha (Tueller *et al*. 1979). A great range of variability in plant species composition exists in this type. While much of this variability is a function of climate and soils, the complex nature of change can also account for much of this variability (Tausch 1998a); perhaps best summarized by Tausch and others (1993) with the statement that there is no "natural vegetation".

In a study of pinyon-juniper overstory relationship to understory plant biomass, Pieper (1990) found that increasing or decreasing tree canopy cover had species specific responses, with some understory species increasing in biomass and others decreasing. In a similar study of canopy influence on understory vegetation, Wilcox and others (1981) found significant preferences of individual species to microsite conditions related to open-area and understory physical and chemical conditions. Specific to pinyon-juniper systems, the influence of tree duff and litter can significantly influence the presence and abundance of understory species (Everett *et al*. 1983).

Fire is an instrument of change that can also provide insight into the comment that there is no "natural vegetation". While average fire frequencies are difficult to assess, a fair body of evidence points to a definite historical occurrence of fires in pinyon-juniper woodlands (Tausch 1998a). Functionally, fire is theorized to have limited tree dispersal and subsequent conversion of surrounding vegetation types. Today, however, the introduction of livestock has created additional gaps in the continuity of fuels, which together with the practice of fire suppression, has allowed for an expansion of these woodlands. In addition to an expansion in total area covered, the structure of existing woodlands is theorized to have changed, with the competitive vigor of mature trees eliminating many understory species (Barney and Frischknecht 1974, Wright *et al.* 1979). As a further complication, the invasion of exotic species, mainly cheatgrass (*Bromus tectorum*) can lead to a fire promotive state. This occurs where cheatgrass establishment is sufficient to form a continuous fuel layer that, with a fire source, re-burns at frequent intervals, eliminating trees from some sites and significantly altering the vegetative composition (Tausch 1998b, West *et al*. 1998).

Traditionally, a primary focus in the management of pinyon-juniper woodlands has been to maintain the productivity of herbaceous and shrub species for livestock and wildlife (Tausch & Tueller 1995) with fire frequently identified as a method for achieving such goals (Arnold *et al*. 1964, Wright *et al.* 1979). Fire is used to increase sprouting of target, fire adapted species and reduce competition of undesirable species. However, as pointed out by Everett and Ward (1984), burns are too often conducted without sufficient information on probable post-fire response. A large portion of this neglect, according to Johnson and Miyanishi (1995), is that ecologists and managers fail to take into consideration the behavior and subsequent ecological effects of fire. In pinyon-juniper communities, Wright *et al.* (1979) identified the response of understory species to fire as the largest void of information.

In natural systems, fire effects can display variability on both large and small scales; as stated by Bond and Van Wilgen (1997), this variability is a focus ripe for study. As a large scale example Turner *et al.* (1997) found significant effects of fire size and pattern on early succession following wildfires in Yellowstone National Park. On a smaller scale, Cave and Patten (1984) found differences between shrub subcanopy and interspaces in a Sonoran desert community. In a different study, temperatures, recorded with the use of thermocolor pyrometers placed in prescription fires, were shown to have extensive spatial variability (Hobbs and Atkins 1988). The nature of fire limits meaningful analysis without some consideration for the spatial variability of fire effects. In pinyon-juniper woodlands, the variability of growth related factors existing between tree canopy and intercanopy microsites can be significant (Davenport *et al.* 1996).

This study was designed to examine the short-term effects of prescription fire on the understory vegetation of a pinyon-juniper woodland through comparison of a control burn plot and adjacent unburned area on a microsite level. Specific objectives were: 1) to characterize 1-year post-burn response of all understory plant species with respect to changes in density (all plants) and cover (grasses and annuals only); and 2) explore microsite recovery patterns.

Methodology

The study site is the Head Chaining Prescribed Burn, an 1,800 acre unit located in the Clover Mountains of Lincoln County, Nevada, and managed by the Bureau of Land Management. Elevations are between 1750 m and 1900 m, with the existence of two distinct zones: 1) a mesa top region with shallow soils, a shallow caliche subsurface soil layer, gentle slopes, and small to moderately developed trees, and; 2) a series of small ravines with variable depth soils, a limited caliche layer, moderate slopes, and moderate to well developed trees. Over the past 69 years average maximum temperature of 21.5 C, minimum temperature of 2.2 C, and total annual precipitation of 23.2 cm have been recorded for Caliente, 14 kilometers northwest of the study area at 1500 m elevation.

During the early 1960's the study unit was treated by mechanical chaining, but follow-up burning was never conducted. Site improvement objectives identified by current land managers included: 1) reduction of decadent browse species by 65% and the creation of a mosaic pattern with 70% burned in the target area; 2) the reduction of young pinyon (*Pinus monophylla*) and juniper (*Juniperus osteosperma*) trees under five years old by 75%; 3) the reduction of big sagebrush (*Artemisia tridentata*) canopy by 50-65%; 4) the increase in resprouting of key desired plant community species, and 5) the raising of mature pinyon-juniper canopy by four to six feet.

Burning was conducted in September of 1997. Approximately one third of the planned acreage was treated before further burning was halted due to fire intensity and fuel consumption levels outside prescription guidelines. Post-fire observations showed almost complete consumption of fuels throughout the treated portion of the study area and surviving, scorched trees only along the burn margins or in rare, isolated islands within the fire perimeter. Data obtained for the five-month period prior to field sampling show approximately average temperatures, but precipitation totals were twice normal.

All post-fire vegetation sampling occurred between June 15-21, 1998. A total of 50 sample trees (25 control and 25 treatment) were chosen for study, with tree selection based on proximity to randomly located points. Sampling was stratified with 60% of the sample points on mesa sites and the remaining 40% on ravine sites. Transition zones were excluded to avoid confusion and reduce variability. Selection of sample trees was further restricted by conformance to an estimated size distribution in order to avoid potential differences due to tree canopy size. The size distribution limited sample trees to previously determined size classes, based on stem diameter 15 cm above ground level. Average burn tree basal diameter was 15.6 cm with a range from 9.0 to 28.1 cm, and average control tree basal diameter was 16.0 cm with a range from 3.35 to 29.9 cm. Sample tree selection was further limited to specimens that shared no more than 50% of total interspace regions with adjacent trees. The determination of canopy drip-line of burned trees was made by extrapolation of the remaining skeletal tree.

Following the selection of sample trees, basal cover (the area of a plant that intersects the ground) of all grasses and density of all living higher plant species was recorded by use of a 50 cm by 20-cm frame. For frequently occurring grasses a smaller 10-cm x 10-cm frame was used (Elizinga *et al.* 1998). A total of 24 sampling points for each tree over 12.5 cm stem basal diameter were recorded, with eight samples at equal distant vectors away from the tree bole (45° degree increments beginning at 0°). These subsamples were repeated at each of three microsites defined as follows: 1) the canopy drip-line; 2) mid-canopy (50% of drip-line); and 3) the tree interspace (150% of drip-line distance). For trees with stem basal diameters less than 12.5, sampling data was taken at every other vector, the 12 missing sample frame data points being treated as missing values (Figure 1). Drip-line and tree interspace recordings that shared canopy cover with adjacent trees were noted and also treated as missing values.

The experimental design was set up as a 2 (burn vs. control) x 3 (microsite) factorial, with 25 replicates and eight subsamples per replicate. This design allowed for a total of 1200 sampling points, but points treated as missing observations reduced total sampling to 941 observations. Density and cover data were analyzed in this design using SAS macro, MXANOVA, and mac-call files Mxqlqls1.sas, Mxqlqls2.sas, and Mxqlqls3.sas (Fernandez 1998). Total species richness, defined as the total number of different plant species occurring in any given sampling frame, was assessed with the addition of a macrosite factor (mesa vs. ravine). The level of significance was chosen as $p<0.05$, with all treatment Least Square Mean comparisons accomplished by using a relatively conservative Tukey-Kramer test (Neter *et al.* 1996); all further mention of the word significance falls under the definition herein provided.

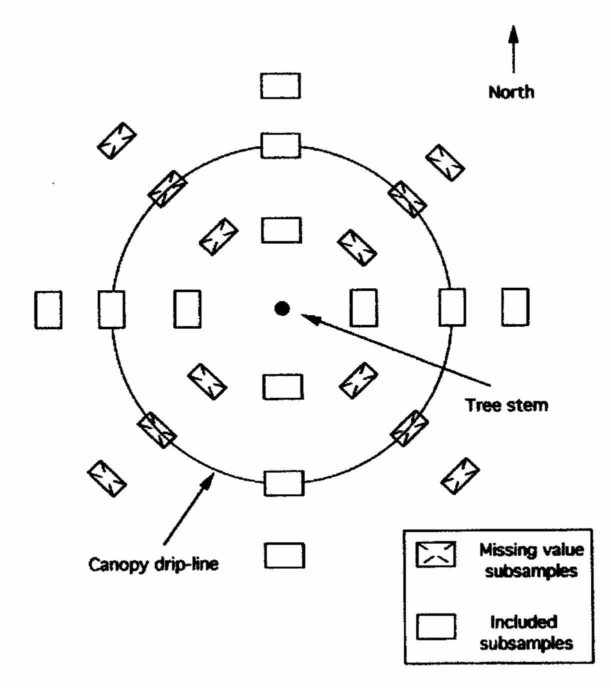

Figure 1. Representative subsampling diagram for tree under 12.5 cm basal diameter

Results

Graminoides

A total of eight grasses were identified to species level in control and burn plots combined, six of which were perennial grasses (Table 1). With the exception of Indian ricegrass (*Achnatherum hymenoides*) no perennial species displayed burn plot increases in cover over control plots. The most prevalent species of this group, bottlebrush squirreltail (*Elymus elymoides*), showed a highly significant ($P>0.0001$) burn decrease. The second most prevalent perennial grass species, prairie junegrass (*Koeleria macrantha*) also displayed a significant decrease ($P>0.0007$). No other individual perennial species difference was significant at the $P<0.05$ level, but as a group, the basal area cover decrease for all perennial grass species combined was significant ($P<0.0001$). The average distributions of pre-burn basal cover of perennial grasses were approximately even, with 1.22%, 1.17%, and 0.70% observed at the three microsites from the tree outward. Post-burn analysis of microsite response showed significant ($p<0.0001$) subcanopy and drip-line microsite decreases, but no significant interspace decrease.

Table 1. Mean basal area percent cover of perennial grass species.

Species	Control	Burn
Agropyrum cristatum	0.0310	0.0000
Achnatherum hymenoides	0.0052	0.0399
Elymus elymoides (a)	0.5708	0.0985
Koeleria macrantha (a)	0.3371	0.0125
Poa fendleriana	0.0178	0.0000
Poa secunda	0.0888	0.0314
Unidentified perennials	0.0127	0.0050
All Species Combined (a)	1.0346	0.1413
Subcanopy (a)	1.2265	0.0055
Drip-line (a)	1.1759	0.2488
Tree interspace	0.7015	0.16949

Species with significant ($P<0.05$) burn vs. control differences using Tukey-Kramer adjustment are followed by the letter (a)

Two different annual grass species occurred in both burn and control plots, six weeks fescue (*Vulpia octoflora*) and cheatgrass. The response of these annual grasses was quite different than that of perennial species. Six week fescue, a native species, had increases in both percent basal cover (from 0.0174% to 0.0909%) and density (from 0.0208 plants/m^2 to 0.1029 plants/m^2), although neither increase was significant at the $P<0.05$ level. In contrast, significant differences were observed for the non-native cheatgrass, the most common of all plant species sampled in both control and burn plots (Table 2). A significant ($P<0.0001$) decrease in the density of burn plot cheatgrass plants occurred, with no observed recovery in the subcanopy burn microsite, a significant ($p>0.0057$) decrease at the drip-line microsite, and a significant ($p>0.004$) decrease in the tree interspace microsite. Despite the density reduction, total basal cover of cheatgrass showed an overall increase of 75% in burn plots, carried by a 230% increase ($p>0.0133$) in the tree interspace microsites. On average, basal cover of individual cheatgrass plants in burn plots was more than 13 times larger than cheatgrass growing on control plots.

Table 2. Mean cheatgrass percent basal area cover and density per m^2.

Microsite	Control		Burn	
	Cover	Density	Cover	Density
Sub-canopy	0.571	9.495	0.000	0.000
Drip-line	1.341	26.110 (c)	1.567	1.434 (c)
Interspace	1.972 (d)	37.996 (e)	4.519 (d)	6.315 (e)
Total Response	1.2947 (a)	24.533 (b)	2.0285 (a)	2.583 (b)

Significant differences ($P<0.05$) between burn and control with Tukey-Kramer adjustment are indicated by similar letter.

Non-grasses

A total of 16 annual and 55 perennial/biennial non-grass species were sampled in control and burn plots combined. Of this total, 35 species (32 perennials and three annuals) were observed only in control plots and eight (five perennials and three annuals) were observed only in burn plots; the remaining 28 species (17 perennials and 10 annuals) were present in both control and burn plots. Further discussion in this section is limited to the 26 most frequent of these species and genera with average combined burn and control densities greater than 0.02 per m^2.

Significant burn related decreases were noted in three of the six most common annual species, birdsbeak, (*Cordylanthus* sp.) ($p<0.0088$), narrowstem cryptantha, (*Cryptatha gracilis*) ($p<0.0058$), and slender phlox, (*Microsteris gracilis*) ($p<0.0165$), although the later two were present in burn plots. Three other non-grass annual species, pinnate tansymustard (*Descuriana pinata*), wedgeleaf draba (*Draba cuneifolia*) and shy gilia (*Gilia inconspicua*), showed no significant change between treatments. Only one annual species, coyote tobacco (*Nicotiana attenuata*), showed a significant ($p<0.0001$) increase in density as a result of burning. Coyote tobacco was not recorded in any control plots and, at 0.1459 plants per m^2, was the most common annual non-grass species inventoried in burn plots.

While a mixed response of perennial non-grass species was observed, most significant burn related changes were decreases (Table 3). The species and species groups showing significant decreases are James' cateye, (*Cryptantha cinerea*) ($p<0.0141$), (*Eriogonum* spp.)

Table 3. Mean density response of common perennial non-grass species

Species	**Mean Density Response**	
	Control	**Burn**
Argemone munita (a)	0.0000	0.0211
Artemisia nova (a)	0.0323	0.0000
Artemisia tridentata (a)	0.1827	0.0000
Astragalus spp. (a)	0.0354	0.0054
Balsamorhiza hookeri	0.0224	0.0053
Chaenactis douglasii (a)	0.0449	0.0000
Chrysothamnus viscidiflorus	0.0305	0.0108
Cryptantha cinerea (a)	0.0409	0.0000
Eriogonum spp. (a)	0.1310	0.0000
Heliomeris multiflora	0.0814	0.0683
Ipomopsis congesta	0.0252	0.0000
Linum lewisii	0.0208	0.0255
Lotus plebius	0.0490	0.0980
Lupinus argentus	0.0293	0.0250
Penstomen confusus	0.0258	0.0270
Penstomen linarioides	0.1254	0.0951
Phlox austromontana	0.0224	0.0039
Phlox longifolia	0.0250	0.0409
Physaria chambersii (a)	0.1507	0.0101
Sphaeralcea ambigua (a)	0.0000	0.1751

Density values are averages per m^2. Species with significant ($P<0.05$) burn vs. control differences using Tukey-Kramer adjustment are followed by the letter (a).

($p<0.0093$), Douglas' pincushion, (*Chaenactis douglasii*) ($p<0.0015$), (*Astragalus* spp.) ($p<0.0357$), big sagebrush, (*Artemisia tridentata*) ($p<0.0002$), black sagebrush, (*A. nova*) ($p<0.032$) and Chamber's twinpod, (*Physaria chambersii*) ($p<0.0019$). Perennial species with no significant difference between control and burn plots include Douglas' rabbitbrush (*Chrysothamnus viscidiflorus*), Nevada goldeneyes (*Heliomeris multiflora*), Hooker's balsamroot (*Balsamorhiza hookeri*), ballhead gilia (*Ipomopsis congesta*), silvery lupine (*Lupinus argentus*), flax (*Linum lewisii*), longbrack trefoil (*Lotus plebius*),Owen Valley penstemon (*Penstomen confusus*), toadflax penstemon (*P. linarioides*), Coville desert phacelia (*Phlox austromontana*), and longleaf phlox (*P. longifolia*); all of these species with burn plot presence displayed some level of resprouting. Two other species, desert globemallow (*Sphaeralcea ambigua*) and flatbud pricklypoppy (*Argenome munita)* showed significant ($p<0.0001$ and $p<0.0498$) increases in burn plots. Neither of these plants was observed in control plots and the more common, desert globemallow, had the highest burn plot density of all perennial species.

Due to their relative abundance and growth form difference from mature shrubs, seedling response was measured separately for Stansbury's cliffrose (*Purshia stansburyana*) and sagebrush (*Artemisia* spp.). Most of the Stansbury's cliffrose seedlings were observed in dense clumps originating from rodent seed caches, while the sagebrush seedlings tended to be scattered around or under the canopy of associated vegetation. No significant change for Stansbury's cliffrose was observed, but a significant ($p<0.0002$) decrease occurred for sagebrush seedlings in burn plots.

Species Richness

Control plot vegetation showed no significant species richness differences among the three microsites, although the response on mesa macrosites was significantly ($p<0.0001$) lower than that on ravine macrosites. Within burn plots, macrosite was not significant but the species richness in sub-canopy microsites was significantly ($p<0.0124$) lower than the tree interspace response. In an overall comparison, burn effect, macrosite, and microsite were all highly significant ($p<0.0001$) as was a burn-macrosite interaction ($p<0.0003$) (Table 4).

Discussion

Graminoides

The most notable burn related changes among grass species occurred with cheatgrass. Past research has shown fire to reduce germinability of cheatgrass considerably, with a direct correlation to fire intensity (Young *et al*. 1976). While fire intensity or fuel loading was not calculated, the microsite pattern of post-fire cheatgrass response indicates the existence of spatial variability in fire effects, displaying an increased density response outward from the tree stem. Studies of understory fuels and vegetation have shown duff depth and litter layer in a decreasing gradient from the tree stem outward (Wilcox *et al.* 1981, Everett *et al.* 1983) along with corresponding changes in fire effects along a similar gradient (Bentley and Fenner 1958). Higher fire intensity in these areas of heavier fuel accumulation caused a reduction of germinable cheatgrass seed as well as other seed of non-adaptive species.

In contrast to the subcanopy microsite, the tree interspace microsites had significantly greater basal cover of cheatgrass, even though densities were much lower. A few different

Table 4. Species richness response by macrosite-microsite-burn combination.

Treatment Combination	Total Species Response per 0.1 m^2*
Burn-Mesa	0.7338 (a)
Sub-canopy	0.3619 (b,c)
Drip-line	0.8518
Tree interspace	0.9880 (b)
Burn-Ravine	1.1329 (a)
Sub-canopy	0.4219 (d)
Drip-line	1.2831
Tree interspace	1.3657
Control-Mesa	1.6230 (a)
Sub-canopy	1.4000 (c)
Drip-line	1.7140
Tree interspace	1.7544
Control-Ravine	2.9281 (a)
Sub-canopy	2.5690 (d)
Drip-line	3.1944
Tree interspace	3.0205

Values followed by similar letters are significantly different (p<0.05) using Tukey Kramer adjustment.
*Statistical results are based upon subsample values, not the actual species richness response listed.

factors could have contributed to this increase. First, the competitive nature of cheatgrass for soil water in conjunction with an elimination of competition would have a significant effect (Young and Evans 1978, Melgoza *et al.* 1990, Whitson *et al.* 1996). Additionally, an increase in available forms of nitrogen common after fire would have contributed greatly to the vigor of cheatgrass.

Perennial grasses showed a significant reduction in cover consistent with early stage successional trend studies (Arnold *et al*. 1964, Barney and Frischknecht 1974). These generalized successional trend studies indicate a quick (5 to 6 year) recovery of most perennial grass species, assuming a representative population of perennials survive the fire treatment. Significant decreases of certain perennial grass species were observed in our burn plots. All such species were present in burn plots. Given this response, no contradiction of previously published studies appears evident. Still, an implied assumption of the rapid recovery of perennial grass species is that competition from surrounding plants does not limit availability of soil resources. Considering the significant reduction of perennial grass cover in the subcanopy and drip-line microsites and the strong recovery of cheatgrass cover in the tree interspace microsite, this assumption may be questionable. Melgoza and Nowak (1991) determined that cheatgrass is a successful competitor and partially reduces the root systems of native species. Cheatgrass competition can significantly reduce plant water potential of native species, with evident competitive effects 12 years after a fire (Melgoza *et al*. 1990). While these results do not indicate a significant long-term change, the dominance of cheatgrass and its influence on recovery of native vegetation is a factor that could influence directional trend of the plant community (Tausch 1998b).

Non-Grasses

Pinyon-juniper woodlands exhibit a short-term dominance of annual species occurring within 1-3 years of disturbance (Arnold *et al.* 1964, Barney and Frischknecht 1974). Ward and Everett (1984) pointed out that the annual stage can be skipped if succession has already eliminated annuals from the stand. Data from our study indicate a relative response: compared to perennials, a much greater proportion of annual species carry over to burn plots. Still, the density response of annual species showed many more statistically significant declines than increases. While literature on species specific declines is often limited, the positive fire response observed by coyote tobacco has been well documented. Baldwin and Morse (1994) found that, in certain soils, the positive response of coyote tobacco was related to increased germination cue and nutrients. For other soils, the high response was more strongly related to a reduction of inhibiting soil chemical properties caused by overstory vegetation.

A typical fire response is for an immediate reduction in the number of perennial species. Burning has been frequently identified as a method for reducing dominance of trees and unpalatable woody species such as big sagebrush (Arnold *et al*. 1964, Wright *et al.* 1979). The reduction in dominance of woody species was evident as no living pinyon or juniper trees or live mature sagebrush were observed in burn plots. Six other perennials were significantly reduced in burn plots. The assumption was made that physiological and/or morphological traits of these species would have made them susceptible to fire effects. Bond and vanWilgen (1996) suggest that specific plant characteristics that raise susceptibility to fire include higher moisture content in exposed tissue, poor vigor, susceptible age or phenological state, or poorly insulated perennating tissue. A review of the post-fire perennials present at this study site suggests a strong reliance on survival of underground perennating tissue for species specific survival (Wright *et al.* 1979).

The significant response of both flatbud pricklypoppy and desert globemallow indicates the presence of some fire or disturbance related mechanism. In a study specific to *Spharalcea ambigua*, Page *et al.* (1966) attributed increases to a stimulation of seed reserves in response to the fluctuating temperature extremes and increased nitrate levels that are both common following fire. Considering the lack of any other regular naturally occurring disturbance in most pinyon-juniper sites, fire may very well be necessary for the site-specific continuance of species such as desert globemallow and coyote tobacco.

Species Richness

Species richness was significantly reduced as a result of burning. The significant short-term reduction in species richness in sub-canopy microsites after burning may help to explain the varied disjunctions as a result of differences in fire intensity mentioned by Humphrey (1984). Likewise, the microsite approach to monitoring vegetation response to fire provides insight into site-specific responses and subsequent knowledge based management steps.

Conclusion

The microsite approach employed in this study provided a convenient means for elucidating the spatial variability of fire effects on vegetation. This spatial variability is most evident among the grass species. The subcanopy microsite pre-burn basal cover of perennial grass species was highest at 1.2265%, but in burn plots was the lowest at 0.0055%. This highly significant drop was attributed to perennial grass species susceptibility to the higher fire intensity that occurred in the subcanopy microsites. This greater fire intensity was not quantified but inferred through field observations and previous research that have shown greater fuel accumulations under trees (Wilcox *et al.* 1981, Everett *et al.* 1983) and associated fire effects (Bentley and Fenner 1958).

The response of cheatgrass to burning is of great concern to land managers due to its fire-promotive potential and competitive vigor with most desirable species. On a microsite scale, cheatgrass had a very strong response in the tree interspace region. The long-term effect of this response is uncertain. Perhaps the only concrete management directive that seems evident from this study is that, if any post-fire cheatgrass reduction measures were desired, they could be focused on the tree interspace regions since the sub-canopy response was negligible. Similarly, if any post-fire vegetation was desired in order to establish target species or to reduce short-term erosion, effort could be focused on the tree sub-canopy microsite as this area had the lowest response of species and least competition from cheatgrass.

The density to cover relationship of cheatgrass is of particular interest. Even with a density response 9.5 times higher in control plots than in burn plots, the basal cover response of burn plots was significantly higher. The contribution of above normal precipitation totals during the growing season following burning would have contributed to this robust response, but such is the stochastic nature of a post fire environment (Whelan 1995). Whether or not the robust cheatgrass response can lead to significant changes or to a new threshold or stage described by Tausch (1998b) is yet to be determined.

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Temporal and Spatial Dynamics of Pre-EuroAmerican Fire at a Watershed Scale, Sequoia and Kings Canyon National Parks

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Abstract

Our understanding of fire as a process in the complex landscapes of the southern Sierra Nevada prior to Euro-American settlement is poor relative to contemporary ecosystem processes. Information primarily consists of fire return intervals from a relatively small number of sites that does not adequately represent past dynamics of fire at a watershed scale. To obtain a better landscape perspective on fire history I used a network of sites located throughout the coniferous zone in the East Fork watershed of the Kaweah River drainage and reconstructed fire regime attributes such as area burned annually and frequency patterns relating to aspect, vegetation type, and elevation. Initial analysis suggested striking differences in the fire regime between north and south aspects with differences strongest at mid-elevations. Fire return intervals on north aspects were less than half that observed on south aspects. Estimates of annual area burned, derived from Thiesson polygons, also showed considerable variability. During certain years fire extended throughout much of the drainage and in a some cases adjacent watersheds also. Pattern and variability in annual area burned were strongly influenced by aspect and annual climate variation. They underscored the importance of climate and topography as controllers of spatial and temporal patterns of fire occurrence. The patterns also suggested strong linkages between fire and ecosystem dynamics with important implications for resource managers restoring fire in the southern Sierra.

Introduction

Historically, fire played a key role in the dynamics of most Sierra Nevada ecosystems (Kilgore 1973), shaping ecosystems temporally and spatially. In much of the mixed-conifer zone, fires were primarily non-stand replacing surface fires (Kilgore and Taylor 1979; Warner 1980; Pitcher 1987; Caprio and Swetnam 1995), although exceptions exist (Caprio et al. 1994). The ignition source of fire prior to Euro-American settlement is usually attributed to lightning or Native Americans. Beginning with Euro-American settlement between 1850-1870, fire regimes in the Sierra Nevada changed dramatically, with sharp declines in fire return intervals in most plant communities (Kilgore and Taylor 1979; Warner 1980; Swetnam et al. 1992; Caprio and Swetnam 1995). Factors contributing to the initial decline include the loss of anthropogenic ignitions and heavy livestock grazing following Euro-American settlement that reduced herbaceous fuels, particularly at lower elevations, and effectively limited fire spread (Caprio and Swetnam 1995). Active fire suppression efforts, begun in the early twentieth century, further decreased or eliminated surface fire as an ecological influence in most Sierran ecosystems. This change in the fire regime led to unprecedented fuel accumulation, structural and composition changes, and resulted in an increased probability of widespread severe fires in many plant communities (Kilgore 1973).

The National Park Service's mission mandates that the agency "protect and preserve" natural resources, which includes restoring and maintaining natural ecological processes. Restoring fire as a process is an important component in this mandate. Beginning in the 1960s land managers began to reintroduce fire within Sequoia and Kings Canyon National Parks (Bancroft et al.1985). While the original emphasis was fuel reduction, the ultimate goal is to restore attributes of past fire regimes in park ecosystems. This has led to a need to better understand the character of these regimes, and how they functioned at both local and ecosystem wide scales. Additionally, information about the range of variation in these attributes is important in understanding landscape level ecosystem processes and in designing and implementing ecologically sound fire management objectives. Unfortunately, most attributes of fire regimes prior to Euro-American settlement are poorly documented particularly in areas with a high frequency surface fire regime.

The Parks have recently begun to utilize and integrate current knowledge about past fire regimes into a GIS framework to provide information for ecologically sound management and for optimizing burn program planning (Caprio et al. in press; Caprio and Graber 2000; Keifer et al. 2000). Various models used in this GIS analysis are directly or indirectly derived from the fire history data (tree-ring analysis of fire scars) and historic fire records (obtained from mapped fires). Using this information to summarize fire return intervals (FRI) in various vegetation types, an "ecological needs" model was developed to produce a map of "fire return interval departures" (FRID). The model highlights areas that have deviated from their historic fire regime following Euro-American settlement. The FRID model is a dynamic and valuable decision support tool that integrates ecological information used to prioritize areas for initial treatment with prescribed fire, assists with scheduling successive burns, helps provide economic accountability, and evaluates progress towards achieving landscape-level ecological goals (Caprio and Graber 2000). The quality of the FRID output is dependent on the quality of the fire history information used in its calculation.

A number of fire history investigations, utilizing fire scar records from trees, have been carried out in or near the Parks over the last three decades (Kilgore and Taylor 1979; Warner 1980; Pitcher 1981, 1987; Swetnam et al. 1992; Swetnam 1993; Caprio et al. 1994; Caprio and Swetnam 1995; Swetnam et al. 1998). They provide important ecological and management information, such as documenting changes in fire frequency resulting from Euro-American settlement, illustrating relationships between fire and forest structure, and showing the interaction of climate and fire. As the breadth of areas and plant communities sampled from throughout the park has expanded, so has our knowledge and understanding about the underlying characteristics and variability of the pre-Euro-American fire regimes. As a result, we are not only becoming more aware of the complex patterns and relationships of past fire regimes but also of the variability in the quality of our knowledge for different vegetation communities and locations within the Parks. Inadequate site replication frequently results in overly simplistic interpretations of past fire regimes at the landscape level. A recent analysis suggests that good quality information is only applicable to about 26% of the Park's vegetated area (Caprio and Lineback in press). For example, one facet of past regimes that has been poorly studied is the effect of aspect on past fire occurrence and spread patterns although its influence on contemporary fire behavior and spread patterns is well documented (Agee 1993; Pyne et al. 1996). Several fire history investigations from other regions of North America have suggested there may be shifts in FRI by aspect (Allen et al. 1995; Laven et al. 1980; Taylor and Skinner 1998; Quanfa et al. 1999).

This study's objective was to obtain well replicated estimates of FRI to assess intra- and inter-site variation within specific vegetation types and aspects throughout a watershed that could then be projected with some confidence to larger land units to reconstruct fire size estimates. This paper presents preliminary details about the fire history sampling that has been conducted using a network of sites throughout a watershed with diverse topographic features and vegetation. Results will eventually answer a variety of questions associated with fire regime attributes–such as frequency and size– and how these vary across a landscape by aspect, elevation and vegetation type.

Study Area

Sampling was conducted in the East Fork drainage (Figure 1), one of five major drainages comprising the Kaweah River watershed, which historically flowed west into the Tulare Lake Basin

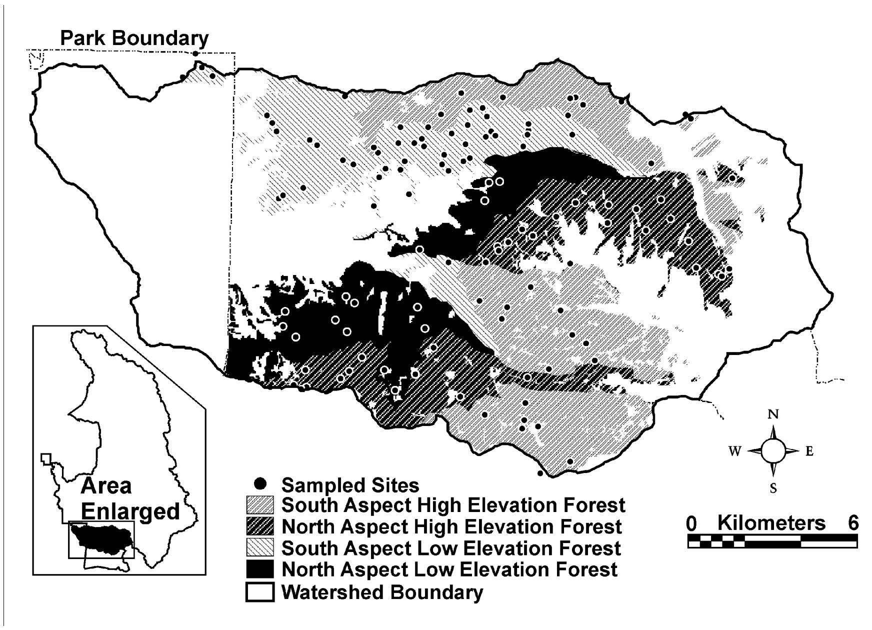

Figure 1. East Fork watershed of the Kaweah River drainage showing distribution of sites within the coniferous forest belt (inset shows watershed location within the Parks). Forest vegetation is categorized by elevation (high and low divided at 2286 m) and aspect (N >285° to <106°, and south 106° to 285°).

in the southern San Joaquin Valley. Terrain is rugged with elevations ranging from 874 m (2,884 ft) to 3,767 m (12,432 ft). The drainage, 21,202 ha (52,369 ac) in size, is bounded by Paradise Ridgeto the north, the Great Western Divide to the east, and Salt Creek Ridge to the south. Major topographic features in the watershed include the high elevation Mineral King Valley, Hockett Plateau, the Horse Creek subdrainage, the high peaks of the Great Western Divide, and the Oriole Lake subdrainage.

The elevation gradient from the foothills to the higher peaks is exceptionally steep with rapid transitions between vegetation communities. About 80% of the watershed is vegetated with most of the remainder being rock outcrops located on steep slopes and at high elevations. Three broad vegetation zones dominate the watershed: **foothill**s (485 to 1,515 m) composed of annual grasslands, deciduous oak, evergreen woodlands, and chaparral shrubland; **conifer forest** (1,515 to 3,030 m) with ponderosa pine (*Pinus ponderosa* Dougl.), lodgepole pine (*P. contorta* Dougl. var Murrayana Englm.), giant sequoia *(Sequoiadendron giganteum* [Lindl.] Buchholz), white fir (*Abies concolor* Lindl. & Gord.) and red fir (*A. magnifica* Murr.) forests; and the **high country** (3,030 to 4,392 m) composed of subalpine forests with foxtail pine *(P. balfouriana* Jeff.), alpine vegetation, and unvegetated landscapes. Vegetation is dominated by red and white fir forests with pine and foothill vegetation of somewhat lesser importance. Ten named sequoia groves are found within the drainage and include the large Atwell, East Fork, and Eden Groves. Portions of the Atwell and Oriole Lake Groves were logged at various times over the last 100 years before being incorporated into the Parks. Since 1990, 2,272 ha (5,611 ac) have burned within the watershed, most of this associated with the Mineral King Risk Reduction Project (Caprio 2000).

The climate is distinctly Mediterranean with cool moist winters and warm summers with little rainfall, although seasonal summer thunderstorms occur sporadically at higher elevations. Precipitation increases with elevation, from 1,515 to 2,424 m on the west slope of the Sierra, to about 102 cm (40 in) annually, then decreases at higher elevations and to the east (Stephenson 1988). Substantial snow accumulations are common above 1,515 m during the winter. Total annual precipitation during the period of record has varied from 30 to 130 cm at Ash Mountain in the foothills and from 38 to 214 cm in Giant Forest at a mid-elevation location.

Euro-American settlement of the area began in the 1860s with extensive grazing, minor amounts of logging, and scattered mineral exploration. Mineral King was the site of a brief speculative mining boom in the 1870s and 80s but little actual ore was recovered. However, a 25-mile long road was constructed into the area in 1878 that provided relatively easy access. Limited logging occurred in the Atwell Mill area in the late 1880s. Sequoia and General Grant National Parks were legislated in 1890, originally with the intent of protecting sequoia groves from logging, but have been expanded to include much of the surrounding rugged, high mountains and some foothills areas (Dilsaver and Tweed 1990). The Mineral King area of the drainage remained under the jurisdiction of the US Forest Service until 1978. Higher elevations of the watershed receive considerable recreation use while most lower elevations are seldom visited.

Methods

Sample sites were located throughout the drainage in the conifer forest belt across all aspects and elevations. Sites were selected to provide good spatial coverage within the drainage. Site locations were constrained by accessability and availability of fire history material. They were located in homogeneous areas with uniform aspect, topographic position and elevation without barriers to fire spread. Areas sampled were small, generally less-than one hectare, to avoid the influence of area on frequency estimates (Arno and Peterson 1983). Samples from multiple trees were usually collected at each site. This strategy has been shown to provide the best record of past fire occurrence (Kilgore and Taylor 1979). It considers each site a single replicate, with individual trees as subsamples, from which a composite fire interval is calculated (Dieterich 1980). Replication is important since every scarred tree will not have a complete record of each fire or may have lost portions of the record in subsequent burns or from decomposition. Within site sampling was usually less intense at higher elevations because a complete record could be obtained with fewer trees since a FRI were longer and forest turnover rates slower (Pitcher 1981). Partial cross-sectional samples were collected using a chainsaw, primarily from dead trees (snags, stumps, or logs) although a few samples were collected from living trees to ascertain twentieth century fire history. Samples from living trees were removed as partial sections (Arno and Sneck 1977) or collected with an increment borer (Sheppard et al. 1988). Position coordinates (UTM) for all samples were recorded and topographic, vegetation, and fuels data collected at each tree sampled. Tree positions were averaged to obtain site coordinates.

Dendrochronological techniques (Stokes and Smiley 1968) were used to determine the calendar year of occurrence for each fire event, visible as a lesions in a catface, and in most cases the specific position within a ring (season) in which these events occurred (Stokes 1980; Ahlstrand 1980; Caprio and Swetnam 1995). Crossdating allowed temporally explicit fire dates to be obtained permitting spatial patterns of burns to be reconstructed.

Area burned annually prior to Euro-American settlement was reconstructed using Thiesson polygons (Davis 1986). Each irregular shaped polygon represents the area around a point (the sample site), in a field of scattered points, determined by Euclidean distance that is closer to that point than any other point. The resulting field of polygons represents the most geometric compact division of area, given the specific arrangement of points. It provides a valuable tool for quantifying and portraying spatial patterns over a landscape. This approach is commonly utilized for rainfall gauging networks when stations are not uniformly distributed and strong gradients occur (Dunne and Leopold 1978)–both characteristics of the network of fire history sites sampled in the East Fork. Polygons and polygon area were determined around the center point of each fire history site using ArcView 3.1 (ESRI 1997) and compiled to create maps of area burned annually from 1700 to 1860. Polygon border delineation was constrained by watershed boundaries and by aspect so a polygon located in one category would not overlap any area in another category.

Fire return intervals and reconstructed patterns of fire size across the landscape were examined by elevation (high/low) and aspect (north/south). Low and high elevations were separated at the transition zone between *Abies concolor* and *Abies magnifica* at 2286 m (Dennis 1999). South aspects were defined as aspects from 106° to 285° and north as >285° to <106° while level areas

(<5% slope) were classed with south aspects (Caprio and Lineback in press). Areas smaller than about 2.5 km^2 were considered to be micro-aspects and categorized as being the same as the surrounding macro-aspect since fire occurrence would be strongly influenced by the overall macro-aspect in which they were imbedded. This was assumed because fire spread and not ignition source was considered the most important influence on fire occurrence on any specific unit of land.

Results and Discussion

Fire History Data

The fire history network is currently comprised of 124 sites with 544 individual trees sampled. Sites were located throughout the watershed's 12,991 ha coniferous forest belt (Figure 1) with 82.4 % falling within 1000 m of another site. The data set included two giant sequoia sites previously collected in the Atwell Grove by Swetnam et al. (1992) and data from trees sampled by Pitcher (1987) supplemented material collected at two sites near Tar Gap. Sites were categorized as being in one of four categories; low-north, high-north, low-south, and high-south. Area in each of these categories was roughly equivalent, differing by less than a factor of two (Table 1). Greater sampling density on portions of the south aspect was a result of attempts to ascertain optimal sampling density in an area of easy access. Mean number of samples collected per site was 4.4 (SD=2.4). Sampling intensity generally varied inversely with elevation. Fire history at higher elevation sites, with longer FRI, could be reconstructed with fewer samples. Fire dates have been determined at 92 sites and form the basis for the current fire return interval estimates and annual area burned reconstruction. A total of 255 dated samples were used in this analysis with over 2050 individual fire scars dated. A total of 304 fire events were recorded between AD 1400 and AD 1995 although fire dates extend back to 284 BC at the sequoia sites (Swetnam et al. 1992). The last fire of significant size occurred in 1889 (recorded at five sites) with 1994 the most recent fire. Most large 20th century fires were recorded by the sampling but spatial information was poor because recording trees had been destroyed by fire, a result of heavy fuel accumulations and severe fire following decades of unnatural fuel accumulation. This highlights the need to collect this historic record before it is lost.

Table 1. Number of sites and size of area by aspect and elevation category within the 12,991 ha conifer belt in the East Fork drainage.

	North (>285° to<106°)		South (106° to 285°)	
	Number of Sites	Area (ha)	Number of Sites	Area (ha)
High (>2286 m)	31	3,088	29	4,360
Low (<2286 m)	15	2,664	49	2,775

Site sample depth varied through time and was used to determine a cutoff date for data analysis. Number of sites recording fire events peaked in about 1800 on both north and south aspects (39 and 52 respectively). Sample depth declined prior to 1700 (28 and 34 sites respectively) on both aspects so analysis was restricted to the post-1700 record.

Temporal Attributes

Fire frequency declined following Euro-American settlement (about 1860) with nearly complete cessation of fires by 1900. This is typical for sites in the southern Sierra Nevada (Kilgore and Taylor 1979; Swetnam et al. 1992; Caprio and Swetnam 1995; Swetnam et al. 1998). Comparison of master fire chronologies showed aspect difference among sites from similar elevations with differences strongest in lower elevation (1776 to 2165 m) conifer forest (Figure 2). For the period from 1700 to 1850 the mean FRI on south aspects was 9.1 yr (N=11, SD=4.7 yr) in contrast to 31.8 yr (N=13, SD=11.4 yr) on north aspects. Differences were less marked between the upper elevation aspects. Within aspect differences were also apparent at local scales and may be the result of variation in vegetation type, specific topographic characteristics of a site, and the influence of elevation. Future analysis will look at these factors in more detail when the full data set for the watershed is developed.

The master fire chronologies also highlight fire years recorded at a large number of sites on both aspects, suggesting fires of widespread occurrence. Years recorded at the greatest number of sites were 1777, 1829, and 1841. Some fire events were only recorded at single sites suggesting locally confined fires. However, because the fire scar record was a conservative recorder of past fire events, the number of sites recording particular events should be viewed as a minimum estimate of fire occurrence. The relative differences among sites probably reflect actual variation in FRI across the landscape.

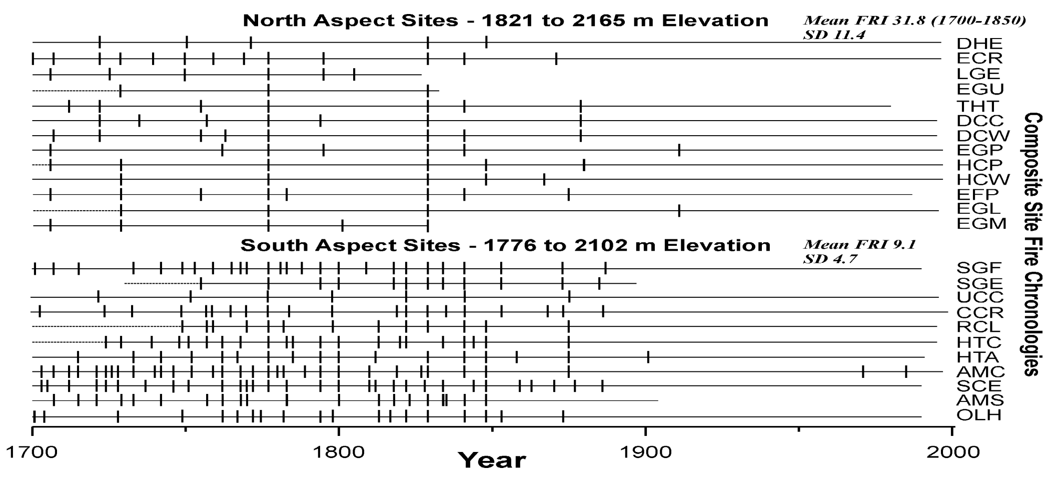

Figure 2. Composite fire interval chart for sites located at low to mid-elevations on north and south aspects. Each horizontal line represents the composite period of record for a specific site and short vertical lines indicate the fire events recorded by site.

Striking differences were also apparent between north and south aspects when the relationship between elevation and fire frequency was examined. Data from the south aspect showed a strong inverse relationship between elevation and fire frequency (Figure 3b). This pattern was similar to the relationship found when fire histories were reconstructed across an elevational transect on a south aspect in the Giant Forest area (Caprio and Swetnam 1995). However, the strength of this pattern did not extend onto the north aspect where I observed only a weak relationship between elevation and fire frequency (Figure 3a). This difference exists even though the same general vegetation types occur on both aspects.

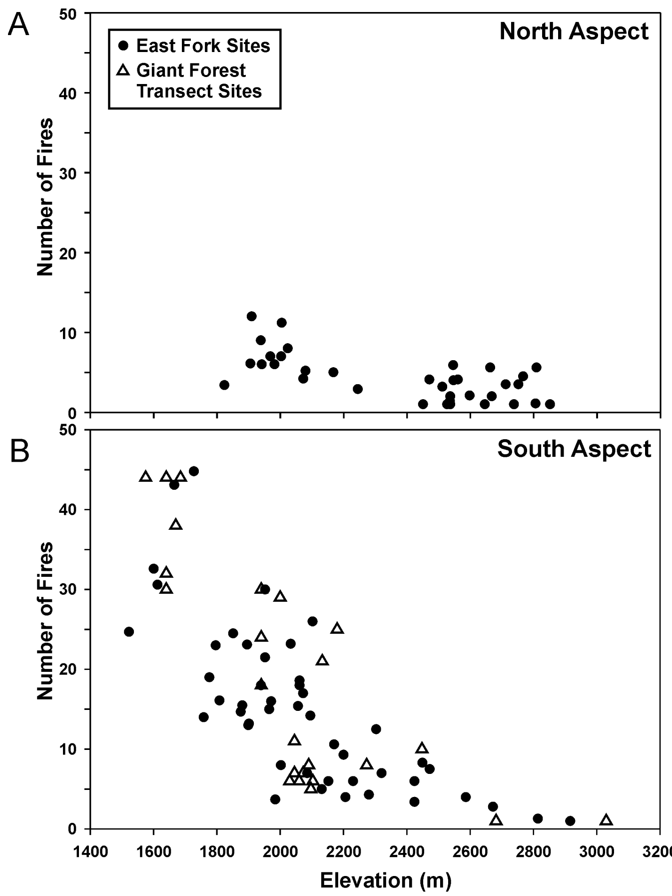

Figure 3. Relationship between number of fire events recorded at a site and elevation by aspect. A strong inverse relationship exists on south facing slopes (b) with a much weaker relationship on north slopes (a). For comparison, fire data from an elevational transect located on a south aspect near Giant Forest are also shown (Caprio and Swetnam 1995, Caprio unpublished data).

Spatial Attributes

Definite spatial patterns of past fire occurrence have emerged as the network of sites throughout the drainage has been collected and Thiesson polygons developed (Figure 4). Mapping

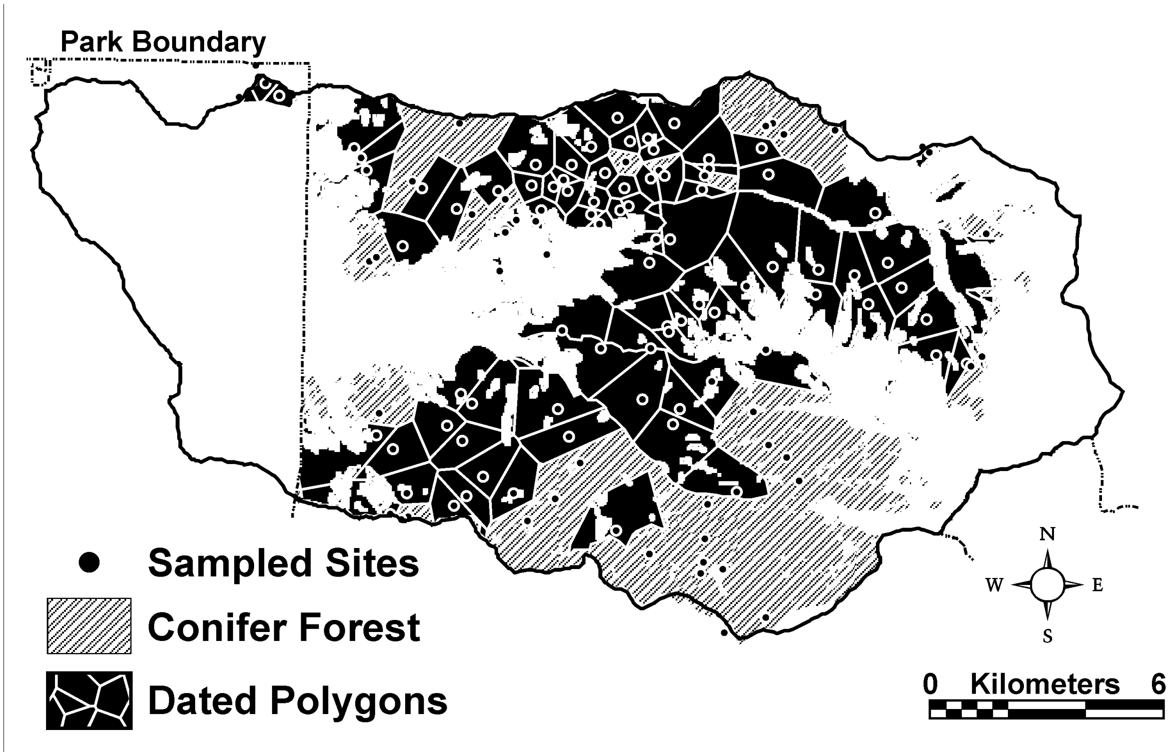

Figure 4. Thiesson polygons constructed around all sites where fire history chronologies have been reconstructed. Current analysis is based on the data derived from this set of sites.

indicated patterns of area burned could be reconstructed over the landscape with some reliability (Figure 5). This was significant because obtaining estimated size of pre-Euro-American fires has generally been restricted to ecosystems where stand replacing fires occur and fire-initiated seral age classes can be dated and mapped (Heinselman 1973; Romme 1982; Barrett 1994). In ecosystems with high frequency surface fire regimes the complex structural differences in the forest vegetation does not permit this type of mapping (Arno and Sneck 1977). However, the sampling strategy used in the East Fork indicates that by sampling a network of sites across a landscape, area attributes can be reconstructed with some accuracy.

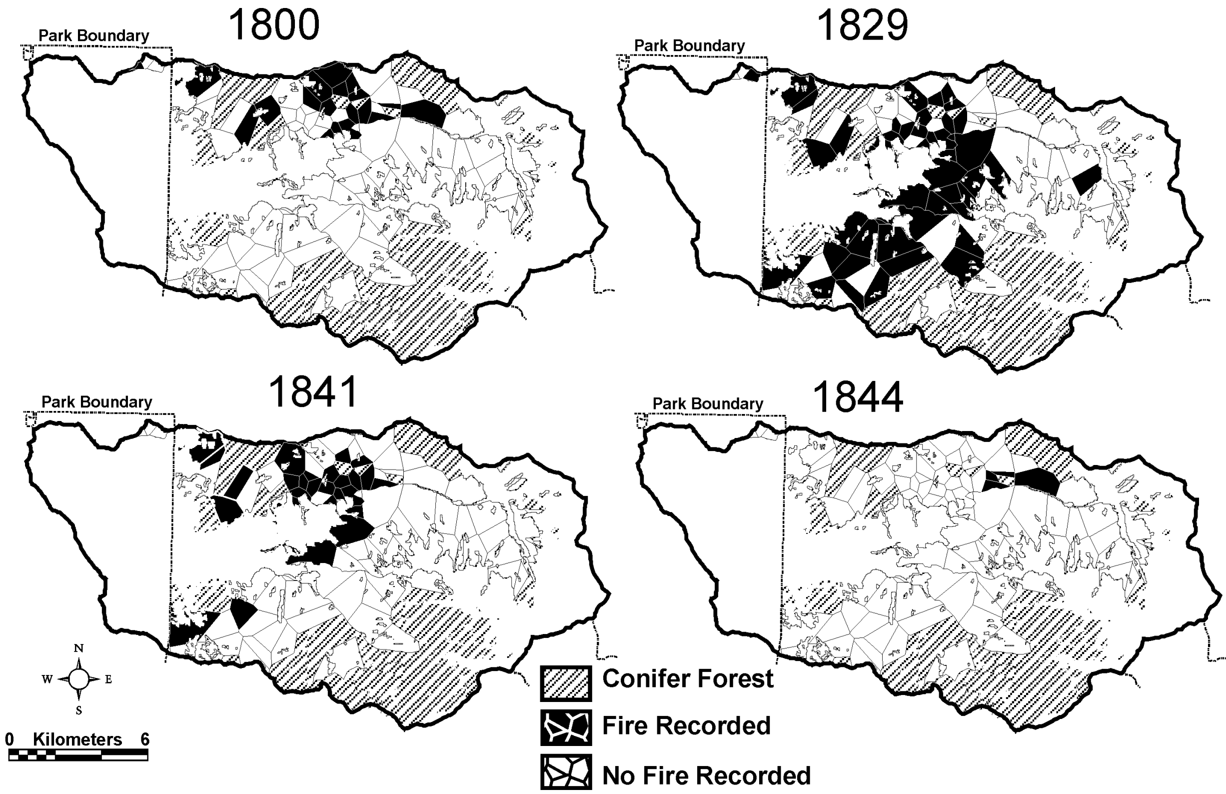

Figure 5. Reconstructed spatial pattern of area burned during four typical years based on polygons from currently dated fire history chronologies. The maps illustrate particular patterns of fire occurrence across the landscape.

While this approach provided an estimate of past fire size a variety of limitations were encountered. For example, the resolution of the final burn maps was commensurate with sampling intensity, and while rough estimates of past fire size may be obtained, precise locations of burn boundaries could not be determined. However, the approach was not biased against areas where understory burns have occurred as are fire size estimates from stand origin maps (Johnson 1992). Additionally, the distribution of point estimates over the landscape generally represented a minimal area burned by a particular fire or fires in a given year. Thus, evidence of a fire scar in a particular year was a definitive record of fires occurrence, while the lack of a scar could be the result of the area not burning or the fires passage leaving no record–either trees were not scarred, scars were subsequently lost, or a sample with the scar was not collected–even though a fire occurred. Fire size estimates from single or isolated sites probably have the greatest error. Size estimates should improve as any particular fire event is recorded at more sites. In contrast, fire events recorded at single sites could overestimate fire size since a small fire might be projected onto a larger polygon. Future sampling of a network of sites across an area where contemporary fires have been mapped would provide considerable insight and validation of the use of Thiesson polygons for fire size reconstructions.

Using the Thiesson polygons a chronosequence of maps showing annual area burned was reconstructed for the years 1700 to 1899 (mean polygon size 104.1 ha, SD=79.8, median 97.6). Several fire years illustrated spatial patterns of reconstructed area burned with a variety of patterns apparent (Figure 5). The 1800 fire date indicated fire was confined to a well defined area on south aspect slopes within the main drainage. Estimated burn area, based on the sum of the polygons, was approximately 1,300 ha. In contrast, burn areas for 1829 and 1841 indicate widespread burns (~4,209 and ~1,938 ha respectively) and indicated areas burned in both the main East Fork drainage and the Horse creek drainage. The spatial pattern of these two burns suggested either conductance across lower elevations in the drainage through vegetation that is currently chaparral and evergreen forest or possibly multiple ignitions. In contrast to these widespread burns, in 1844 only a small cluster of four sites recorded what was probably a single relatively small fire event (~255 ha).

Also of interest were comparative maps showing extent of area burned in 1873 and 1875 after Euro-American settlement. The reconstructed area for the 1873 burn (16 polygons) showed that fire was centered on the central portion of the Atwell Grove while the map for 1875 (26 polygons) indicates burning took place predominantly to the east and west of this area and on lower slopes of the north aspect. Overlaying the polygon burn maps from these two burns indicated that they were essentially mutually exclusive with both dates co-occurring at only one of forty-two sites. This indicated fuel recovery in two years (1873 to 1875) was not sufficient to permit extensive reburning. John Muir (1878) also made interesting historical observations about the 1875 fire that verifies its occurrence in chaparral vegetation between the south and north aspects of the watershed. He wrote that while traveling in the Atwell area he watched this fire burn intensely up-canyon through chaparral vegetation and enter the sequoia grove where intensity decreased markedly.

A time series of reconstructed annual area burned showed considerable year-to-year variation from 1700 to 1899 (Figure 6a). Fires were recorded during 106 of the 200 years with years of particularly widespread fire (>2,000 ha based on polygon reconstruction) in 1777, 1829, and 1848. Reconstructed burn area in eleven other years exceeded 1,000 ha. Average area burned annually within the watershed was 320 ha or about 2.4% of the coniferous forest area. The distribution of area burned annually within the watershed from 1700 to 1899 showed an inverse J shaped distribution (Figure 6b). Most fires were small (mean size 462 ha) with a few years when extensive fire occurred (~3,720 ha in 1777).

Fire sizes or area burned in some years probably exceeded fire area estimates based solely on the East Fork Study area. Fire in these years may have been a single widespread burn or a complex of burns across multiple watersheds. Fire history sampling in adjacent Kaweah River drainages (South, Middle, and Marble Forks) showed many years in common to the East Fork (Swetnam et a. 1992; Caprio and Swetnam 1995; Swetnam et al. 1998; Caprio unpublished data). Prominent fire dates in the record from the South Fork (south of the East Fork) included 1777, 1812, and 1841 whereas in the Middle Fork/Marble Fork areas (north of the East Fork) common dates include 1777, 1795, 1812, 1829, 1841, and 1873. These common fire years also occurred across the landscape at the regional scale (west slope of the southern Sierra Nevada) and appear related to annual climatic variation (Swetnam 1993; Swetnam et al. 1998).

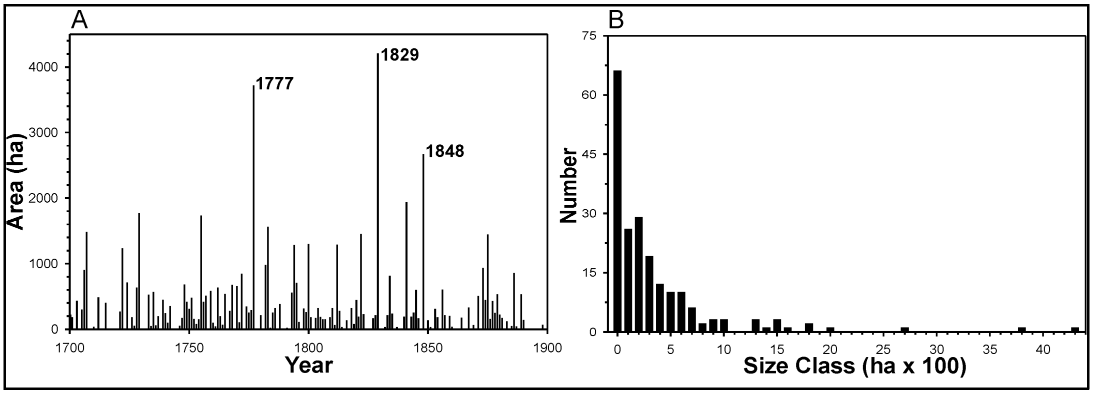

Figure 6. Area burned annually within the East Fork watershed from 1700 to 1899 (a). Area values were reconstructed from the 92 polygons for which fire chronologies have been developed. In most years area burned was small (mean 310 ha) with a few years when large areas burned. (b) Distribution of total area burned by size class for the period from 1700 to 1899 showing that small fire years were responsible for most of the area burned within the watershed.

Aspect and Elevational Influences

Dramatic differences in area burned annually became apparent when data were separated by elevation and aspect (**Figure 7**). They were greatest between lower elevation north and south aspects

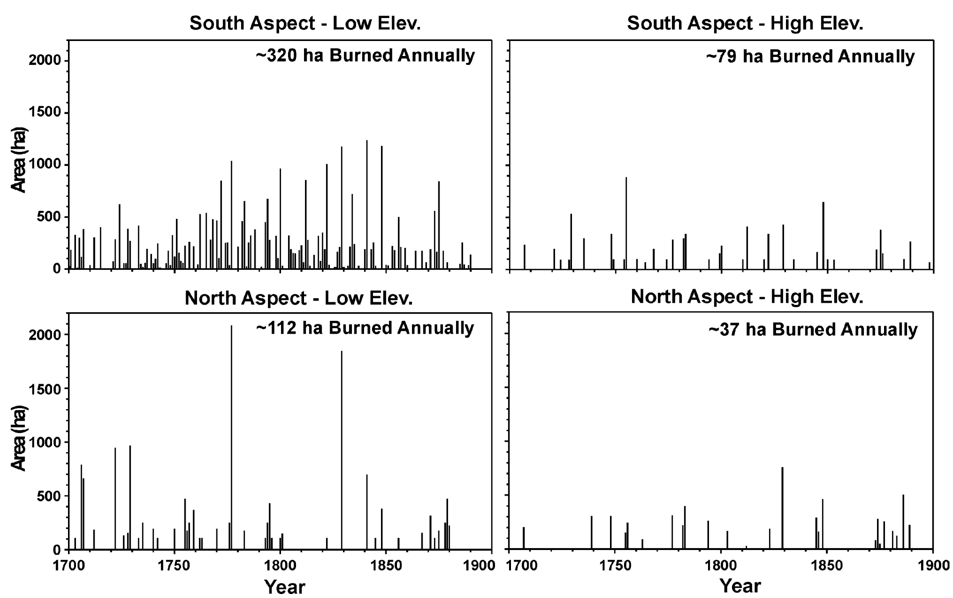

Figure 7. Area burned annually by topographic category, high / low elevation and north/south aspect. Differences between aspects were greatest between the lower south aspect and other three aspect categories while south aspect showed greatest fire frequency.

and less striking between higher elevation north and south aspects. Annual area burned on south aspects was generally small but was regularly interspersed with years when moderately large fire years occurred. This pattern contrasted markedly with patterns from the north aspect and from higher elevation south aspects where the incidence of fire occurrence was lower and more irregular. However, among the latter groupings subtle differences also existed. For example, on the lower elevation north aspect the longer FRI were punctuated by years when large area burned (1777 and 1829).

Specific mechanisms driving these patterns are unknown but are likely associated with the interaction of fuels, vegetation, and climate. While reliable data on fuel and forest structure conditions prior to Euro-American settlement do not exist, there are proxy records of precipitation derived from dendroclimatic studies. A preliminary examination of the climatic relationship between past rainfall and fire event occurrence on north versus south aspects showed patterns suggesting at least partial mechanisms. The relationship was investigated by overlaying fire dates onto a time series of an indexed residual tree-ring chronology. The chronology was developed from climatically sensitive low elevation trees in the Kaweah drainage and has a significant correlation with January to April (winter) rainfall (Caprio unpublished data). When all fire events were considered, fire years on the south aspect (Figure 8a) co-occurred with almost equal likelihood to years with less than average ring widths (50 dry years) or years with greater than average ring widths (75 wet years) while fire years on the north aspect (Figure 8b) had a stronger overall association with small rings (14 wet versus 42 dry years). However, in years when large areas burned ($\geq$500 ha) fire years on both aspects were strongly associated with small ring widths (on south aspects two wet versus 18 dry years and on north aspects one wet versus 11 dry years). This suggested that both past fire occurrence and size were restricted on north aspects to years of below normal precipitation while on south aspects fire occurrence may have been governed by fuels, with climate largely determining fire size. When the data were further separated by elevation, both upper elevation north and south aspects exhibited burn patterns similar to the overall north aspect discussed above. Future analysis will examine relationships between climate and fire occurrence by specific vegetation types.

These patterns of fire on the landscape are similar to patterns described by foresters observing fire spread over the landscape in the late 19th and early 20th centuries. Show (1922) summarizes these observations and suggests that drought and lack of precipitation for long periods are necessary for extensive fires to occur and that in most years conditions do not permit fires of large size. He reports that fuel moisture differences between north and south aspects provide the mechanism for limiting fire spread in average years. He states "*...we can infer that fires may start every year, that every year they will spread, but that except in very dry years or following dry lightning storms, they may burn out on north slopes, due to the moisture content of the litter, though spreading freely on south slopes. Very extensive fires....are likely to be associated with unusual drought*." The historic patterns described by Show are very similar to patterns reconstructed from the contemporary tree-ring record. Both his observations and the burn patterns from the East Fork suggest that north aspects were effective barriers to the spread of large fires under pre-Euro-American conditions during years of average precipitation. Whether such mechanisms still operate given fuel and vegetation changes over the last 150 years is unclear.

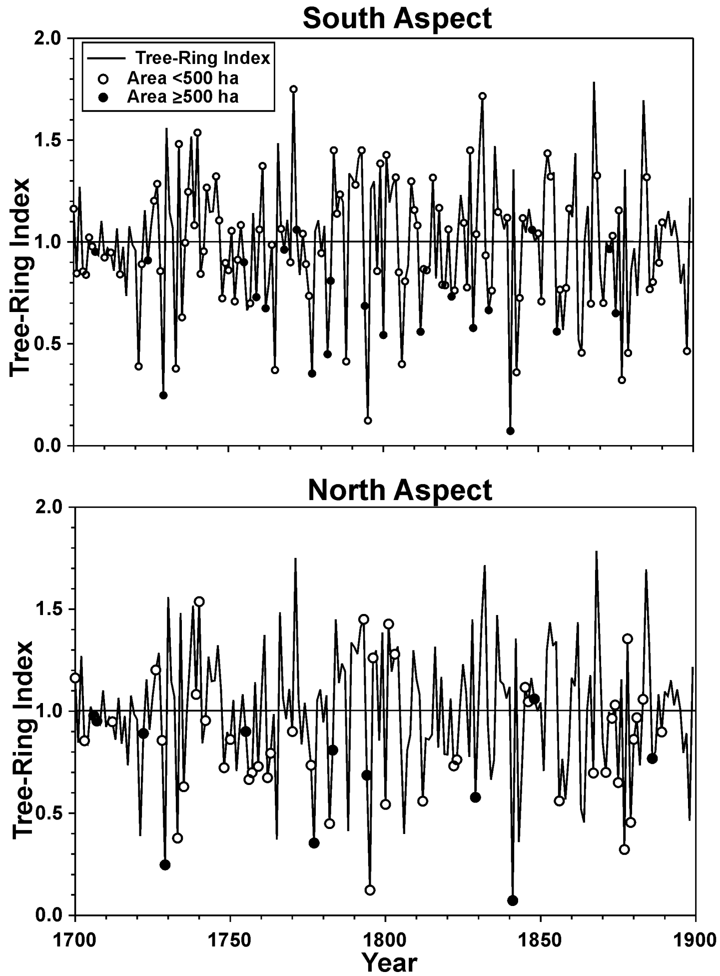

Figure 8. Relationship between fire occurrence and a proxy record of precipitation (drought sensitive standardized tree-ring chronology) on south (a) and north (b) aspects. Open dots show all fire events and solid dots show the relationship if only events ≥500 ha in size are superimposed on the tree-ring indices.

Biotic and Management Implications

Fire regime differences between aspects may have been substantial not only in respect to FRI, but also in relation to biotic effects. The reconstruction of pre-Euro-American settlement FRI on lower elevation south-facing slopes indicated fires were typically light while the more episodic burns on north aspects and at higher elevations were probably of much greater severity. The episodic events would have been related to both fuel conditions (greater fuel loads) and occurrence of fires during dry years. These fires would probably have increased the frequency of gap formation and patch development (such as brush fields), influenced species composition, and created greater spatial and temporal heterogeneity in stand structure. The apparent outcome of fire suppression probably fostered more homogeneous vegetation on all aspects with increased potential for large severe fires occurring throughout an entire watershed or across several watersheds.

Fire regime differences by aspect will also have important implications for management efforts aimed at restoring fire to park ecosystems. Incorporation of the new FRI information into the Park's FRID analysis (Caprio et al. in press) will improve output quality, particularly when analyzed by vegetation type. The data presented in this paper on aspect differences showed that conifer forest on lower elevation south aspects have missed a larger number of fire events than those on comparable north aspects. As a result they have a larger fire return interval departure and have deviated the most from pre-Euro-American settlement conditions. Because these areas may have greater "ecological need" they may become higher priority areas for carrying out management burns than north aspects.

Information about both past fire frequency and annual area burned can also be used to derive targets for fire management planning, and to evaluate how well restoration goals are being met. Current estimates from Sequoia and Kings Canyon National Parks are that on average between 6,100 to 10,000 ha burned annually within the Parks prior to Euro-American settlement (Caprio and Graber 2000). In contrast, the current burn program averages about 1,663 ha burned annually (1985-1999 data), although it is having a significant effect restoring fire to key areas that protect important resources or break up large continuous areas of heavy fuel. While a variety of constraints exist (see Caprio and Graber 2000), a major obstacle to burning at pre-Euro-American levels is the backlog of areas that have missed many FRI where the extremely unnatural fuel loads and dense vegetation makes them difficult and costly to burn with significant smoke production. Until these hurdles can be overcome the Parks' burn program will continue to fall behind. The use of models such as FRID–using the best available fire history data–will continue to help target critical locations for restoring fire as an ecosystem process.

Conclusions

While most investigations of pre-Euro-American fire regimes have focused on patterns of fire frequency, other attributes of the fire regime may be equally important in understanding fire as a process in ecosystem dynamics. The data and analysis in this paper provide not only an estimate of frequency patterns across a large landscape but also a coarse understanding of annual area burned. Both are important in understanding ecosystem processes and dynamics and the scale of the task required to restore fire in this ecosystem.

Within the East Fork watershed considerable variation in FRI were found based on fire history chronologies developed from a network of sites throughout the drainage. The chronologies showed the importance of climate and topography as controllers of spatial and temporal patterns of fire occurrence. The reconstructions also showed (1) differences in mean FRI among sites related to both elevational and aspect, (2) occurrence of common fire years among sites at varying scales, and (3) a connection between fire regime and annual climate variation related to aspect and elevation. Comparisons of FRI between north and south aspects for low-to-mid elevation sites indicated fire was about three times more frequent on the south aspect. There was also a strong inverse relationship between number of fires and elevation on south aspects and a much weaker relationship on north aspects. Additionally, examination of burn patterns by aspect and elevation indicated fire could occur on lower south aspects during almost any year while fire at other locations and large fires across all aspects were related to dry years.

If such patterns are consistent in other drainages they will have important implications for fire and resource managers in terms of planning, anticipating potential fire effects in these areas, and in understanding processes responsible for restoring or maintaining attributes of past vegetation structure and composition. Such information also provides vital baseline data for judging the magnitude and extent of change in park ecosystems over the last 150 yrs and for evaluating whether our fire management accomplishments are meeting NPS mission goals. Eventually, predictive models of general fire regime characteristics across the landscape could be developed based on topographic and biotic influences.

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Fire Management in Yosemite National Park: A Historic Overview and Recent Results

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Introduction

It has been my profound privilege to lead one of the most consistent and progressive fire management organizations into the new millennium. Having led this organization for the last decade, I can reflect on how truly progressive it is by comparing both quantitatively and qualitatively some of the history that has made it so. It is only in retrospect that some of these things can be discussed, because at the time of the events there were thoughts of imminent disaster in the minds of those involved.

As with each fire we manage, there are times when you think that your next paycheck may be sent to some unknown place. This feeling is common when dealing with the unforeseen risk that accompanies a fire program, and each manager from the past who attempted to ignite prescribed fires or manage wildland fires in the wilderness probably had these same fleeting thoughts. The common thread among us all is the same drive that led the research scientists, practitioners, and politicians of days gone by to insist upon a change of culture in the fire community. This culture change is now complete as we enter the next century with a concept that fire is fire and the only difference is how we elect to manage it. The newest federal policy, which applies to all the federal agencies, allows for a varying degree of appropriate management based on analyzed risk. Newer technology allows us to reduce risk by preplanning and analyzing the factors involved instead of simply accepting that risk.

A Brief History

Fire was once unilaterally viewed as an agent of destruction and all agencies declared war against wildland fire (Pyne 1982). Following years of dedicated research and experimentation by prominent researchers like the late Dr. Harold Biswell, National Park Service policy was changed in 1968 (National Park Service 1968). This major change in fire policy recognized fire as a natural process rather than as a menace, and provided an opportunity for individual units to develop plans to utilize fire for resource benefit. The new policy brought about an immediate response from Sequoia and Kings Canyon National Parks (Kilgore and Briggs 1972) and Yosemite National Park (van Wagtendonk 1978). In Yosemite a prescribed fire program was initiated in 1970 and a wildland fire for resource benefit program was initiated in 1972.

As these programs were implemented, there was a natural line of demarcation drawn between some personnel of the National Park Service and other agencies. After numerous decades of active fire suppression it was understandable that some people would not easily give up their belief that fire was ultimately destructive just because of the whim of some influential academic. This was the case in Yosemite National Park at the onset of the program. As fire suppression was deeply seated in the Division of Visitor Protection, the new program was to be just as deeply rooted in the Division of Resource Management. These two divisions had differing philosophies regarding the role of fire. A situation of opposite beliefs between supervisors festered for nearly 20 years despite management support and good progress in the implementation phase. Natural attrition eventually settled these disputes, and a management decision was made to place the programs in a single organizational branch. It is interesting to point out that while these feelings existed, which regretfully ended numerous professional working relationships, the program continued to expand. Great strides were made through the first two decades of the program despite challenges from internal and external sources.

My interpretation of this is that the will to succeed was extremely strong in the people who were implementing the program. The relationship with the other division, though strained, gave them added resolve to push on and grow. Although the fire suppression group would not completely agree with the philosophy of wilderness fire in the early years, they were agreeable with the goals and protection provided by properly executed prescribed burns. Through time, the adversaries began to understand the need for wildland fire managed for resource benefit as well. This understanding was based on continuing research and documentation of fire effects. The effects of fire became a major concern to managers charged with the protection of cultural and natural resources. It likewise became a major emphasis of some of the agency's research community. Not only would additional information provide sound justification to continue the programs locally, it would help provide a rationale for personnel to support the fire program, which I believe did happen. National emphasis was placed on monitoring and documenting fire effects and a national archive was established to start assembling these data.

While the 1970s were an era of experimentation with new programs, the 1980s became a decade for expanded research (Parsons and van Wagtendonk 1996) and additional challenges. Review of programs (Christensen et al. 1987) recommended continued research into new realms including effects on vegetation, building fire histories, long term forest dynamics, fuel dynamics, fire modeling, visitor response, and further investigation into the effects of burning by Native Americans. Earlier studies linked Native American burning to the Sierra Nevada (Reynolds 1959) but the extent and significance regarding change in vegetation character in Yosemite by Native American cultures required and received further research (Anderson and Carpenter 1993, Anderson 1993). A National Fire Effects Monitoring protocol (FMH) was established for the National Park Service during the late 1980s and adopted early in 1990. The intent of this is to provide consistent long term documentation as the program continues in the national parks. This protocol provides documentation of our start point and what is occurring as we continue. It does not tell us where we need to be; a question asked by many of us at this time and surely a question that will need further analysis as we move forward. During the 1980s a significant increase in activity occurred as the organization added positions, expanded fire management zones, and generally grew in confidence. This was a period when the most work was done in Yosemite as shown by Table 1 and figure 1a.

Table 1. Fire use history -Yosemite National Park 1970-1999

	ANNUAL TOTALS				CUMULATIVE TOTALS			
Year	**Wildland Fires**	**Wildland Acres**	**Prescribed Fires**	**Prescribed Acres**	**Wildland Fires**	**Wildland Acres**	**Prescribed Fires**	**Prescribed Acres**
1970			7	664.00	0	0.00	7	664.00
1971			8	1,112.00	0	0.00	15	1,776.00
1972	8	0.31	1	64.00	8	0.31	16	1,840.00
1973	27	56.09	2	192.00	35	56.40	18	2,032.00
1974	22	4,131.71	2	259.00	57	4,188.11	20	2,291.00
1975	20	773.87	7	2,283.00	77	4,961.98	27	4,574.00
1976	35	803.83	13	914.00	112	5,765.81	40	5,488.00
1977	24	149.76	1	20.00	136	5,915.57	41	5,508.00
1978	33	2,485.65	3	4,023.00	169	8,401.22	44	9,531.00
1979	6	78.24	5	2,932.00	175	8,479.46	49	12,463.00
1980	25	6,203.39	5	3,152.00	200	14,682.85	54	15,615.00
1981	39	3,262.39	1	2,450.00	239	17,945.24	55	18,065.00
1982	5	1.17	2	3,120.00	244	17,946.41	57	21,185.00
1983	6	1,660.16	2	2,458.00	250	19,606.57	59	23,643.00
1984	20	1,067.25	3	840.00	270	20,673.82	62	24,483.00
1985	22	3,765.65	6	2,067.00	292	24,439.47	68	26,550.00
1986	8	3,808.11	3	194.00	300	28,247.58	71	26,744.00
1987	40	7,072.98	0	0.00	340	35,320.56	71	26,744.00
1988	43	12,265.00	0	0.00	383	47,585.56	71	26,744.00
1989	0	0.00	11	1,306.00	383	47,585.56	82	28,050.00
1990	21	200.80	9	170.00	404	47,786.36	91	28,220.00
1991	20	1,304.00	7	113.00	424	49,090.36	98	28,333.00
1992	34	576.00	9	950.00	458	49,666.36	107	29,283.00
1993	5	1.80	11	1,073.00	463	49,668.16	118	30,356.00
1994	7	2,147.40	12	1,344.00	470	51,815.56	130	31,700.00
1995	6	815.50	15	399.80	476	52,631.06	145	32,099.80
1996	16	1,469.00	12	1,168.70	492	54,100.06	157	33,268.50
1997	19	127.40	15	5,018.00	511	54,227.46	172	38,286.50
1998	21	200.60	11	2,779.80	532	54,428.06	183	41,066.30
1999	18	14,871.80	14	1,616.60	550	69,298.86	197	42,682.90

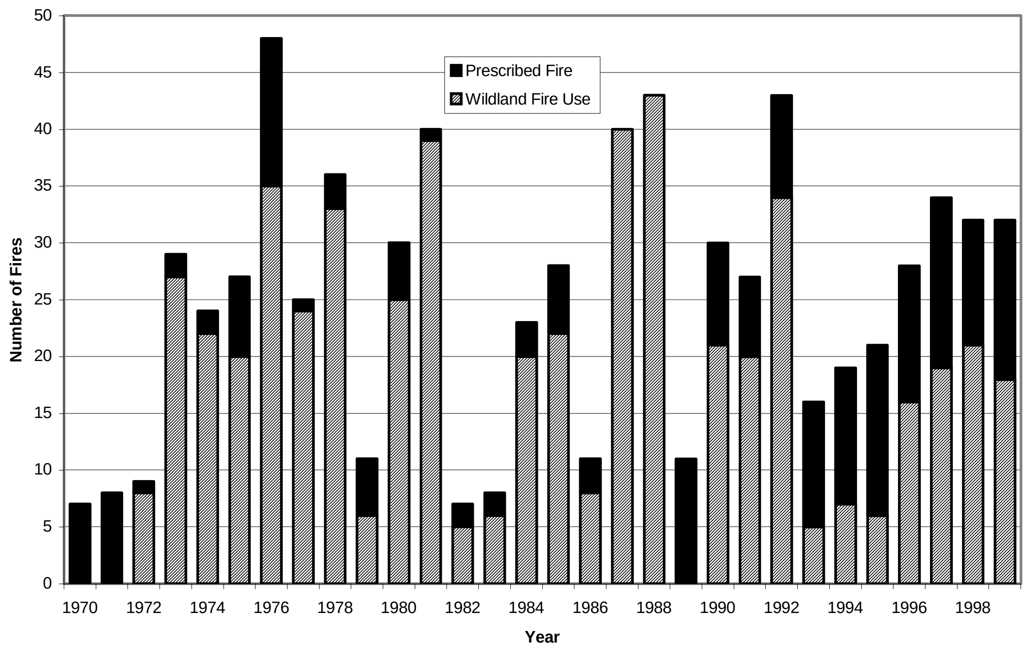

Figure 1a: Fire Use History – Number of Fires by Year

Challenges to the programs began to surface as the activity increased through the 1980s. While prescribed burning and suppression were accepted practices, the value of wildland fire for resource benefit was still not completely accepted. Other agencies had endorsed the practice in their wilderness areas as well but had suspended several regional programs due to unfortunate outcomes which received significant public scrutiny. Requirements for planning and execution became very time consuming and it became easier not to endorse fire use than to complete the necessary paperwork to continue. The risk of dealing with whatever weather you received during these long term situations proved too risky. This often resulted in being out of the traditional weather prescription, which was required as part of the original fire plans. In addition, the availability of firefighters was always a problem with regard to wildland fire for resource benefit since these fires were not considered an "emergency". Largely related to the different funding mechanisms of the federal agencies, there was a perception that fire fighting resources were funded for emergency operations only. To assign them to non-emergency events, when they may be needed for "emergency" wildfires became a serious issue to program managers. It became an excuse, for some, not to support what was largely becoming scientific fact, that the forest required fire to be healthy. This significant problem was not solved until the late 1990s, after yet another policy revision and a funding mechanism change requiring congressional approval.

Smoke became a serious issue in the late 1980s and continued to grow as a problem throughout the next decade (Haddow 1983, Chambers and Duncan 1993). Smoke can easily be seen as the most prominent issue we face in this new century as well. As the populous began to move into the foothills of the Sierra Nevada, several problems surfaced. An increase in population along the forest and park boundaries started to create an urban interface problem very similar to that experienced in some coastal areas of California decades earlier. This increase in people meant elevated concern about deteriorating air quality, since many had moved to the mountains for their clean air. Many were retirees who, having completed a career in the more populated areas of California, were well educated in how to lodge complaints against government agencies but uneducated about fire ecology. To them fire was still a demon to be extinguished immediately. Recall that at this time people had not been through the information exchange that was about to occur between the media, academia, agencies and the public. Their message addressed the need for fire in fire dependent ecosystems.

Perhaps the last major events of the 1980s affecting the evolution of Yosemite's programs was the national realization that fires were not behaving like they did a decade or so earlier. Fuel accumulation and the effects of fire exclusion were finally recognizable by the fire management community and the world. All over the west fires became increasingly hard to put out and covered larger and larger areas. The fire seasons of 1985 in Idaho, 1986 in Eastern Oregon and 1987 in California and Oregon showed this trend. Despite the most modern tools, equipment, high levels of manpower and an extremely effective national mobilization process, these fires burned until winter. Events in 1988 would forever change the perception of wildland fire. In Yellowstone National Park and the surrounding northern Rocky Mountain areas, fires burned for several months and covered distances that truly dumbfounded the most educated of our fire management community. As the decade closed, a resurgence in fire modeling became a priority.

It became altogether clear that the next generation of fire behavior models was needed. The models we had were fine for two dimensional fires, but did not take into account drought effects on large fuels, historic weather events and many other factors that make fire move on a three dimensional scale. Risk analysis was born in the fire management field and we needed better ways to assess risk when dealing with fires that would be managed throughout an entire fire season. This would become a task for the 1990s.

In Yosemite the fire management program became truly integrated in the 1990's when park managers placed all fire management functions in a single organizational branch. The 1988 Yellowstone fires had resulted in special attention being paid to the wilderness fire program and policy was again rewritten to make management of wilderness fires more acceptable. The program was affirmed and education of the public became a national priority if we were to deal with the problems of a national effort to reduce fuels in forest ecosystems. For once all agencies and many private organizations agreed that forest health was at stake, especially in short return interval forest types, which had been the most affected by fire exclusion (van Wagtendonk 1984). Fire programs received additional staffing to address the growing concerns and accelerate the process of restoring a more natural stand structure.

Large wildfires burned the Yosemite in 1990 requiring managers to use interagency crews rather than keeping it within the Park Service. The concept of interagency partnering was finally a reality. Yosemite reached out and was welcomed as a player in the Interagency Fire Community. Why this hadn't happened earlier in the program eludes me, but much of it had to do with the park's endorsement of fire programs that were not widely accepted in California. Fully integrated programs that integrate fire use on a large scale are still not many in this state although acceptance is now much more widespread. Firefighters have been made available and new programs are being started yearly as other agencies get their mandated planning completed.

Special emphasis in the 1990s was placed on dealing with hazard fuels in developed areas so wildland fires would be more manageable if they approached. In 1994 a serious incident involving the fatality of 14 firefighters in Colorado further elevated the need to deal with the "fuel buildup" situation. The term "defensible space" became popular and was given the appropriate priority. All agencies began to focus on reducing fuels to provide for firefighter safety as well as to reduce the intensity of fires. Prescribed fire became more and more the tool of preference for treating large tracts of land to reduce fuel loading. The number of prescribed burns in Yosemite increased but the acreage decreased as smaller protection burns became a focus (Figure 1b). Alternative fuel treatments were experimented in Yosemite to reduce smoke problems. Manual removal of fuels was effective but had to be followed by broadcast burning to meet the resource objectives of exposing mineral soil and removing duff. This treatment was used to quickly open up overgrown areas where visibility and defensibility were key objectives. Thinning and chipping was done on two small units in the valley but proved to be very expensive and slow going. It is now standard in our developed areas to do some thinning and pile burning the year before and to follow up with some pile burning to provide defensibility. It was not until 1993 that the park returned to the wildland with larger landscape-scale burning.

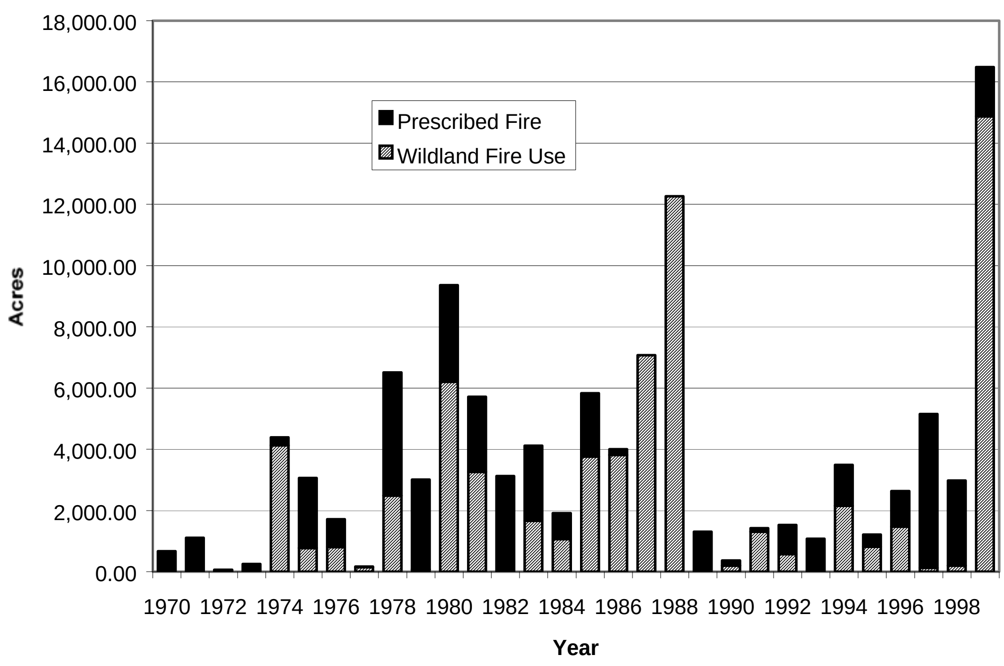

Figure 1b: Fire Use History – Acres of Fires by Year

Wildland fire for resource benefit received added attention as people started to understand the dynamics of fuel accumulation and it relationship to forest health. Following the events of 1994, a new national policy review called for all agencies to endorse all forms of fire management to get the job done. Fire managers began to realize that "appropriate management response" could involve using fire to reduce fuels on larger tracts during an emergency response. Current Incident Management Teams use the mobilized forces to deal with emergency wildfire and the fuel problem surrounding a fire area. Firefighter safety issues and the expanded knowledge of the fuels make it acceptable. Over time this will have a significant long-term benefit as the unnatural fuel buildup from fire exclusion is slowly removed. These concepts were underscored during a review of the management of the Ackerson Fire Complex in Yosemite in 1996. Firefighter safety and wilderness values were seen as the top two objectives in the portion of the fire that was in the park. Additional acres were burned rather then put personnel in a position of danger. What was realized afterward was that very little ecological damage had been done by this, and the land had received a treatment by fire in an area that hadn't seen fire in over 100 years.

The time was right following the events of 1988 for expanded research into fire behavior modeling. The 1990s saw an incredible reemphasis placed on fire behavior. Literally hundreds of models were developed in this decade to assist fire managers do a better job. In Yosemite several of these models were tested on actual fires as they burned in the park, and many are used regularly nationwide. The advancement in computers and information technology has created an environment where development of new models in support of fire management, smoke management and fuels treatment is likely to continue for some time. Coordination with other land management agencies and regulatory agencies has improved dramatically. As the concept of ecosystem health is endorsed by many, the practice of sharing resources, ideas, technologies and facilities will continue to grow.

The Aspen Valley Project

As was mentioned previously, our future is still a mystery. We desire to return the forest ecosystem to a more natural state as far as vegetation type, structure, and fuel loading, but we are not yet clear on the details. Numerous historic descriptions tell us what settlers observed as they visited the forests before the turn of the century. However, no quantitative information is available and these descriptions are vague as to their location and extent. Several papers (Heinselman 1978, Kilgore 1983) discuss and analyze the term "natural" in its application to fire management in parks and wilderness. Unfortunately they fail to describe the "natural" structural attributes of a forest. To quantitatively define this for a given forest type is laborious and still may not be acceptable to all. One concept that I have received concurrence is to revisit sites that have received fire on an interval consistent with the fire return interval of that forest type. Once multiple fire treatments have been applied, whether through suppressed wildland fire, prescribed fire or wildland fire managed for resource benefit, the area should yield a condition closely resembling "natural structure". In line with this thinking we have started to focus on areas that have burned a minimum of three times in three decades. This may not be enough disturbance to replicate a natural condition, since no one knows how many times it will take to remove the effects of fire exclusion.

Site Information

The first area we looked at was the Aspen Valley project which is designated Parkwide Unit #2 (PW-2). Aspen Valley is located in the northwest portion of Yosemite National Park. At 4800 to 6200 feet in elevation is an area that was burned three times. The fire effects plots are located at about 5100 feet. The site of the plots and most of the prescribed burn unit are on principally south-facing slopes. The slopes range from 10-35 percent at the plots. The overstory in the area is principally composed of mature ponderosa pine with a component of mature sugar pine and a few incense-cedars. The understory is typically sparse with thick patches of incense-cedar and white fir reproduction. Manzanita is present but typically sparser than in lower elevation ponderosa pine-bear clover associations. Bear clover comprises most of the vegetative cover with a mixture of low cover herbs and grasses.

Prescribed Fire Effects Plots

In 1983, prior to the first prescribed burn, 20 fire effects plots were established in Burn Unit PW-2. These plots were burned by prescription in October of 1983 and were re-read immediately. The plots were then read each year from 1984 to 1987. These circular point-style plots were archived in 1987, when the NPS developed a new monitoring protocol which in 1990. This new Protocol (FMH) specifies a rectangular style of plot and a different but comparable set of measurements. The unit and plots were burned by wildland fire in August 1990 and again by prescribed fire in November 1998. Parts of the unit were not burned until January 1999 and it is unknown if the plots were burned in 1998 or 1999. The plots were not monitored before, during, or after either of these fires but were relocated in 1999 and re-read in October of that year. Because the plots were not re-read prior to the wildfire in 1990 and the prescribed burn in 1998, the plot data are not as useful as it would be if each set of pre-burn and post-burn measurements were available for comparison. The fire effects program intends to continue reading these plots on a schedule comparable to the schedule defined by the FMH.

Plot Measurements

The circular point style of plot, installed throughout Yosemite in the 1980s, collects five different types of information:

- Basal area and overstory tree data via prism method (BAF 30).
- Seedling and sapling density including height class via one mil-hectare subplot per plot.
- Fuel loading from one 35' Brown's (1974) transect per plot, including fuel and duff depth.
- Vegetation cover and non-living ground cover from four 1 sq. m. subplots per plot.
- Photographs from the plot center to the end of the Brown's transect.

Results

We looked at several components of the plot information for changes or lack of changes associated with the three burns. Forest fuel loading was calculated in tons per acre and also in percentage of the original fuel loading for comparison to prescribed fire goals (Tables 2a&b).

Table 2a. Forest fuels at Aspen Valley in tons per acre.

Fuel Class	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Post (1999)
1-hour	0.20	0.06	0.07	0.10	0.11	0.16	0.06
10-hour	0.69	0.07	0.15	0.20	0.26	0.52	0.37
100-hour	0.52	0.00	0.08	0.23	0.22	0.44	0.97
1000-hour Sound	19.00	17.00	16.30	6.00	6.20	6.20	22.3
1000-hour Rotten	4.90	0.00	6.80	10.10	10.20	10.20	0.80
Duff	33.20	1.80	1.80	2.70	2.10	2.90	3.30
Total Fuel	58.51	18.93	25.2	19.33	19.09	20.42	27.8

Table 2b. Aspen Valley forest fuels: percent of original fuel load by class.

Fuel Class	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Post (1999)
1-hour	100	27	35	49	55	81	32
10-hour	100	10	21	29	38	75	54
100-hour	100	0	15	44	42	85	187
1000-hour Sound	100	90	86	31	32	32	117
1000-hour Rotten	100	0	139	207	208	208	16
Duff	100	5	5	8	6	9	10
Total Fuel	100	32	43	33	33	35	47

Information on litter loading and maximum fuel depth is not presented due to concern over the consistency of these measurements. Seedling and sapling density per ha was also calculated and tabulated by species (Tables 3a&b).

Table 3a. Aspen Valley tree seedlings (0cm to 50cm tall) per hectare

Species	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Post (1999)
White fir	50	0	0	750	0	0	0
Incense-cedar	3,450	0	0	500	32,250	7,400	1,2
Sugar pine	100	0	0	150	100	400	0
Ponderosa pine	300	0	5,100	7,100	8,450	4,450	150
Canyon live oak	50	0	50	50	0	50	150
California black	0	0	0	0	50	0	0
California bay	0	0	0	0	50	0	0
All species	3,950	0	5,150	8,550	40,900	12,300	1,5

Table 3b. Aspen Valley tree seedlings (50cm tall to 15cm dbh) per hectare.

Species	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Post (1999)
Incense-cedar	2500	250	0	0	0	0	0
Sugar pine	350	0	0	0	0	0	0
Ponderosa pine	600	100	0	0	0	0	0
Canyon live oak	150	0	0	0	0	50	50
All species	3600	350	0	0	0	50	50

Overstory individuals were sampled and tabulated by species and dbh size class (15-cm increments) (Table 4a&b). In order to convey maximum information about the distribution of the overstory through the size classes, overstory data are presented in total individuals sampled rather than basal area or stand density. Extensive information was collected on vegetation cover and ground cover. Unfortunately, due to unresolved identification during the 1980s and the need to spend significant time correlating species identifications between the various reads, data on changes in low percent cover herbs and grasses are not presented. However percent cover of bear clover are presented in Table 5. The results of these calculations are discussed individually below.

Table 4a. Aspen Valley overstory: number of individuals sampled by species.

Species	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Post (1999)
White fir	2	1	1	2	2	1	
Incense-cedar	38	33	34	34	34	34	35
Sugar pine	7	6	6	7	7	7	7
Ponderosa pine	96	89	88	89	97	96	101
Black oak	4	4	4	5	5	4	5
All species	147	133	133	137	145	142	148

Forest Fuels.

In ponderosa pine-bear clover associations, fuel accumulations are typically not particularly large. Prior to the 1983 burn, there were 58 tons/acre of total fuel present, principally composed of 33 tons/acre of duff and 19 tons/acre of solid 1000 hr fuels (Table 2a&b). After the first burn, total fuel was greatly reduced from 58 tons/acre to 19 tons/acre, a 68 percent reduction. Duff and rotten 1000-hr fuels comprise most of the 39 tons/acre of consumed fuels (31 ton/acre and 5 tons/acre consumed, respectively). Small woody fuels (1, 10, and 100-hr fuels) were all greatly reduced as well (27 percent, 10 percent, and 0 percent of the original weight, respectively). Between the 1983 fire and 1987, several fuel classes re-accumulated but total fuels remained near post-burn levels. Solid 1000-hour fuels decline as they are converted to rotten 1000-hour fuels; total 1000-hr fuels remain at post-burn levels. Small woody fuels remained low but re-accumulated to near pre-burn levels (81 percent, 75 percent, and 85 percent, respectively).

Pre and post-burn data are not available for the 1990 wildland fire and 1998 prescribed burn, limiting discussion of changes due to these fires to speculation. Fuels were probably reduced by the wildland fire and then re-accumulated until the 1998 prescribed burn. Current total fuel loading is greater than all other post-burn readings, largely due to an increase in 1000-hr fuels. In 1999, solid 1000-hr fuels are at 22 tons/acre, a 17 percent increase over the 1983 pre-burn levels. Overstory mortality induced by the three burns is the most likely cause. 100-hour fuels have also increased to 187 percent of the 1983 pre-burn weight, probably for the same reason.

Seedlings and Saplings

Tree species were counted by species and height class in a mil-hectare plot around the prism point. We split these individuals into seedlings and saplings by height in order to better show the height distribution. We defined seedlings as 0cm tall to 50cm tall (height classes 1 to 3) while saplings were defined as 50cm tall to under 15cm dbh.

One of the general goals of prescribed burns in ponderosa pine-bear clover associations is the reduction of invasion by incense-cedar and white fir. The 1983 burn was extraordinarily successful at reducing the density of seedlings and saplings (Tables 3a&b). Seedlings were eliminated (100 percent reduction) while sapling density was reduced 90 percent. Incense-cedar seedlings were reduced from 2500 individuals/ha to 250/ha by the first burn. No saplings survived to the fall of 1984. Live sapling numbers remained at essentially zero from 1984 to 1999. The live sapling data do not show the dead individuals we found in 1999. Clearly, some recruitment occurred between 1987 and 1999 but was killed by the either 1990 wildland fire or the 1998 prescribed burn. Seedling recruitment is more variable. Seedlings were absent after the 1983 burn but seedling density between 1984 and 1987 was greater than pre-burn levels. There is a surge in ponderosa pine seedling recruitment in 1983 followed by a surge in incense cedar recruitment in 1986. However, despite the increase in seedlings, we can see from the sapling data that survival into the larger size classes is essentially zero.

Unfortunately, we do not have sufficient data to discuss the 1990 wildfire and 1998 prescribed burn in detail. There was a pulse of incense-cedar recruitment in 1999 and a few ponderosa pine seedlings, but reproduction is essentially absent at the one-year mark. Because we lack pre-burn data for 1998, we don't know how many of these seedlings were present prior to the burn. The 1998 burn was patchy and probably permitted the survival of more individuals than the 1990 burn. It is also worth noting that the 1550 seedlings per ha in 1999 represent only 31 individuals counted in 20 mil-hectare plots. Most of these individuals (28) were found in a single plot. Seedlings are essentially not present in the remaining plots.

Overstory.

We chose to present the raw number of individuals sampled rather than the basal area or stand density because our primary interest is in overstory mortality. By showing the number of individuals by species and size class, we see an excellent profile of overstory size distribution and change (Table 4b). Despite two prescribed fires and one wildland fire, this profile shows relatively little change in the overstory. The 1983 burn appears to have killed a few mature incense-cedar and ponderosa pine (size classes 90-105 & 105-120) but otherwise appears to have made little change. In 1999, overstory tree numbers were close to 1983 levels, principally due to large trees growing into the prism circle rather than due to true regeneration in the smaller size classes.

Unfortunately, because the plot methodology does not currently tag individual trees, individual mortality cannot be counted. Variation in the use of the prism may account for small changes in the mature trees sampled. Determining whether a sampled tree died or was simply not counted is difficult without individual tagging or extensive investigation. During the 1999 reading, the methodology was changed slightly to include notes regarding the inclusion or exclusion of individuals previously sampled. If future readings include this information, individual mortality can be examined more closely. However, the data do indicate that the general goal of minimal mature tree mortality was met. The 1983 burn probably minimized mature tree mortality that might have occurred during the 1990 wildfire. However, the site of the plots lacks a substantial subcanopy of white fir and incense-cedar that is present at some other locations in this prescribed burn unit. Visual examination of nearby areas indicates patches of subcanopy regeneration that were killed by the wildland fire or the most recent prescribed burn.

Table 4b. Aspen Valley overstory: total number of individuals sampled by size class.

Species	**Size Class (cm)**	**Pre-burn (1983)**	**Imm. Post (1983)**	**1 yr Post (1984)**	**2 yr Post (1985)**	**3 yr Post (1986)**	**4 yr Post (1987)**	**16 yr Post (1999)**
ABCO	15-30	1	1	1	1	1	1	
	30-45	1			1	1		
ABCO Total		2	1	1	2	2	1	0
CADE	0-15							1
	15-30	1	1	1				
	30-45	2	1	1	2	2	3	2
	45-60	4	4	5	5	5	5	6
	60-75	6	6	6	5	7	7	4
	75-90	10	10	10	9	7	7	11
	90-105	9	7	8	10	10	9	9
	105-	5	3	2	2	2	2	1
	120-	1	1	1	1	1	1	1
CADE Total		38	33	34	34	34	34	35
PILA	15-30	1						
	75-90	1	1					
	105-			1	1	1	1	1
	135-	5	5	5	6	6	6	6
PILA Total		7	6	6	7	7	7	7
PIPO								
	15-30	11	10	10	10	11	10	7

Species	Size Class (cm)	Pre-burn (1983)	Imm. Post (1983)	1 yr Post (1984)	2 yr Post (1985)	3 yr Post (1986)	4 yr Post (1987)	16 yr Pos
	30-45	6	6	6	7	10	10	12
	45-60	5	5	4	5	5	5	10
	50-75	6	5	5	5	6	5	2
	75-90	12	11	11	11	10	10	14
	90-	20	17	17	18	16	16	18
	106-	13	12	11	12	13	12	14
	120-	10	10	11	9	11	11	8
	135-	8	8	8	7	8	11	9
	150+	4	4	4	4	4	3	5
PIPO Total		96	89	88	89	97	96	101
QUKE	0-15				1	1	1	1
	15-30	1	1	1				
	30-45	1	1	1	1			1
	45-60	2	2	2	2	2	2	2
	60-75				1	1	1	1
QUKE Total		4	4	4	5	4	4	5
Grand Total		147	133	133	137	145	142	148

Vegetation Cover.

Due to a need for additional site visits to perform some botanical archaeology, data on change in percent cover of herbs and grasses cannot be presented. Change in percent cover of bear clover is presented because of its tendency to cause small seedling death by shading. The 1983 burn essentially eliminated aboveground bear clover. As expected, resprouting follows the burn and slowly increases the percent cover of bear clover from a post-burn level of 0 percent to 20 percent. The lack of a return to pre-burn levels of 31 percent cover indicates that the fall burn of 1983 was relatively hot. In comparison, the late fall/winter burn of 1998 was cooler and has allowed the more rapid return of bear clover. One year after the 1998 burn, the bear clover cover was 20 percent compared to 2 percent cover one year after the 1983 burn, probably due to differences in maximum soil temperatures achieved during fall and winter burns.

Figure 5. % Vegetative cover of Bear Clover

	Pre-burn (1983)	Imm. Post (1983)	1 yr post (1984)	2 yr post (1985)	3 yr post (1986)	4 yr post (1987)	16 yr post (1999)
Average cover	31.0	0.0	2.3	10.3	11.8	12.3	20.2
90% confidence interval	9.9		1.2	5.4	6.0	6.6	7.0

Conclusion

The results from the Aspen Valley burns give hope that we can return the forest ecosystem to a more natural state. Only through an integrated program of fire management combining wildland fire suppression with prescribed fire and wildland fire use can the objectives of firefighter safety, fuel hazard reduction, and restoration of natural fire regimes be accomplished. We feel we have met those objectives with our program in Yosemite national Park.

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Post-Fire Erosion Control and Vegetation Recovery Monitoring at Point Reyes National Seashore

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Abstract

The 1995 Vision Fire was the most devastating wildfire in Point Reyes in sixty years, with over 12,000 acres burned. Fire suppression relied heavily on dozer line construction. Much of this line was constructed on steep slopes or cut through known noxious weed occurrences. The Burned Area Emergency Rehabilitation (BAER) team, a multi-agency, multi-disciplinary team of resource specialists, was assembled to assess fire damage and fire suppression effects and prepare mitigation recommendations. At the BAER team's suggestion, erosion control measures were implemented immediately. The team also recommended monitoring and mitigation of impacts on vegetation, with particular emphasis on rare plants and noxious weeds. Funding was obtained from FIREPRO and a three-year project was carried out to implement the BAER team's recommendations.

Introduction

In October 1995, the Point Reyes National Seashore (Seashore), north of San Francisco, experienced its largest wildfire in sixty years. The Vision Fire burned more than 12,000 acres of state, federal and private lands in three days (USDOI 1995). Over 95% of the lands burned were within the Point Reyes National Seashore. The fire originated from an illegal campsite on Mt. Vision. Fanned by 30-50 mph winds, the fire spread rapidly through multiple vegetation communities, from ridge top bishop pine (*Pinus muricata*) and Douglas-fir (*Pseudotsuga menziesii*) forests, to coastal scrub and sand dunes at the Pacific Ocean. Vegetation resources were affected to varying degrees as burn intensities varied across the landscape (USDOI 1995).

A Department of the Interior Burned Area Emergency Rehabilitation (BAER) Team was assembled to assist the Seashore in assessing fire and fire suppression impacts that would require mitigation. Seashore staff and other local experts worked closely with the team, providing information and field assistance. Existing pre-fire data on vegetation communities, particularly rare plants and non-native plant infestations, were crucial to acquiring emergency funds to accomplish mitigation measures prescribed by the BAER Team. The Vegetation Management staff at Point Reyes National Seashore was directly involved with three projects funded by FIREPRO: dozer and hand line rehabilitation, non-native plant monitoring, and rare plant recovery monitoring.

Dozer and Hand Line Rehabilitation

Suppression of the Vision Fire resulted in the construction of 23.1 miles of dozer line, 6.4 miles of line constructed by hand tools, 10 safety zones, and 13 drop points and helispots (USDOI 1995). Approximately eight acres of disturbed soil resulted from the construction of safety zones, drop points and helispots. In its soil and watershed resource assessment, the BAER Team predicted "serious watershed deterioration problems" if suppression-related damage was left untreated.

Treatment of dozer and hand lines was prescribed to divert runoff from the lines and to break up concentrated flows. To rehabilitate hand lines, slash and other organic material was placed over mineral soil to stabilize the surface and aid in revegetation (USDOI 1995.) All dozer lines were treated in one of two ways. For areas with slopes of less than 30%, excavators were used to recontour the original slope, to remove berms, and to pull back displaced soil. A hummocked arrangement of soil and interspersed woody material was then created to slow and absorb runoff. Finally, the excavators distributed topsoil and plant material to restore local seed and microorganisms to the site.

Most dozer lines on slopes of greater than 30% were not excavated. These sites were covered with photodegradable erosion cloth. This material must be in direct contact with soil to effectively control erosion. Hand crews prepared the sites by smoothing the soil surface, installing waterbars and removing berms and large woody material. There were several drawbacks to using photodegradable erosion cloth. For example, installation and maintenance were very labor intensive. The plastic netting holding the matting in place was unsightly and did not photodegrade completely. Worse yet, on windy slopes, large pieces of netting broke free, littering the park and occasionally entrapping deer. Lastly, the metal pins used to secure the matting were difficult to remove and often were left behind in the soil.

Hydrologic monitoring in the first and second years revealed rapid vegetation recovery and very little movement of soil on lines rehabilitated by excavators. Monitoring of lines covered by erosion matting revealed that waterbar construction was more effective than the matting in inhibiting soil movement (B. Ketcham, Point Reyes National Seashore, pers. comm.). It was speculated that the steepness of some slopes and high wind intensity at ridgetops caused the lifting up of the matting from its contact with the soil. The matting also appeared to inhibit initial plant regeneration, but visible vegetation recovery at these sites was observed by spring of 1997.

Non-Native Plant Monitoring and Mitigation

The BAER Team predicted that the loss of topsoil on newly disturbed sites would increase potential for the encroachment of non-native, invasive plant species. Heavy machinery used in fire suppression is also capable of introducing non-native seeds from outside sources. Recommendations were outlined for two years of monitoring fire suppression impacts on noxious weeds within the fire area (USDOI 1995.) During the second year of monitoring and mitigation, the need for further monitoring was recognized and funding was awarded for a third year.

Three years of post-fire non-native plant monitoring and removal provided extensive information on the response of a variety of non-native plant species to the Vision Fire and to disturbance related to fire suppression. To facilitate monitoring, all trails, dozerlines and hand lines in the burn were divided into approximately ½ mile segments using ArcView-generated maps. Burn monitors were responsible for regular monitoring of assigned areas and for removing non-native species according to priorities established after monitoring.

Monitoring involved walking the length of each segment and documenting various aspects of vegetation regrowth such as abundance and approximate percent cover of both native and non-native plant species. Populations of non-native species also were sketched on field copies of ArcView-generated maps, which were filed and updated as the season progressed. Pioneer weed populations and species of highest priority for removal were digitally mapped with Global Positioning System (GPS) units.

Monitors prioritized weed removal sites based on habitat type, size of the non-native population, and accessibility of the work site. Sites of controllable weed infestations in bishop pine habitat were given highest priority to encourage the survival of native pine seedlings. Plant phenology generally determined the schedule priorities for control of non-native species. Various thistles were targeted in spring, Australian fireweed (*Erechtites minima*) was targeted in early summer, and pampas grass (*Cortaderia jubata*) was targeted in late summer and fall. Over a three-year period, more than 3,000,000 non-native plants of 40 different species were removed from within the Vision Fire area. Volunteer labor contributed significantly to this total. The numbers of each species removed at each site are maintained in an Integrated Pest Management (IPM) database.

Known locations of two "A" rated noxious weeds, and one "B" rated noxious weed, were disturbed by dozer activity during the fire. According to the California Department of Food and Agriculture, "A" rated noxious weeds require eradication, quarantine or other holding action at the State and county level. "B" rated noxious weeds require intensive control or eradication, where feasible, at the county level. Monitoring of "A" and "B" rated noxious weeds in the burn will continue indefinitely, independent of fire-related funding. Known sites of giant plumeless thistle (*Carduus acanthoides*), fertile capeweed (*Arctotheca calendula*), and distaff thistle (*Carthamus lanatus*) populations were disturbed by fireline construction and are included under these categories. Between 1975 and 1995, giant plumeless thistle (GPT) had been contained to three known sites that were monitored monthly for new growth. In August 1996, a previously undocumented population of GPT was discovered in the Vision Fire area. The population of 12-foot high flowering thistles covered two acres that had been covered by coastal scrub prior to the fire.

Qualitative assessments of areas affected by the fire and suppression activities supported the prediction of increased encroachment by non-native plants. Burned or disturbed areas that occasionally supported scattered occurrences of weeds prior to the fire often supported extremely dense, expansive occurrences after the fire. By late spring of 1996, for example, the plant species composition on many dozer lines was dominated by one or two species of non-native grasses and forbs, monocultures of weeds on otherwise diversely vegetated hillsides. Unfortunately, however, few conclusions can be made on the specific effects of the Vision fire and fire suppression activities on weed establishment and spread, or on the effectiveness of the weed removal program. This is due, in large part, to the following three factors:

- Little or no pre-fire data existed on the botanical resources of concern in the Vision Fire area;

- This project did not involve rigorous quantitative assessment of weed recruitment, spread, or removal treatments; and

- Control plots outside of the burn were not established.

Without quantitative pre-fire data and systematic quantitative documentation of weed parameters both in burned and control areas, conclusions regarding the effects of the fire and project effectiveness must be qualitative in nature. It is not possible, for example, to separate the effects of the fire on non-native plant presence and abundance from the effects of climate and rainfall in the area from 1996 through 1998. The prolonged rainy season brought on by an El Niño system in 1998 resulted in prolific growth and an unusually long flowering period for native plants and weeds alike. It also is not possible to make conclusive statements about the effectiveness of non-native control methods used during this project.

Rare Plant Recovery Monitoring

Rare plant monitoring was conducted for one year after the Vision Fire to document the post-fire recovery of known populations, to document the effects of the fire and fire-related suppression activities on species recovery, and to propose appropriate mitigation measures as needed (Semenoff-Irving 1996). Species composition was determined at each rare plant site according to the protocol outlined in the Western Region Fire Monitoring Handbook (Semenoff-Irving 1996). California Native Plant Society (CNPS) survey data from previous years were used to compare pre- and post-burn density values within populations (Semenoff-Irving 1996). The following six rare plant species were monitored in 1996:

1. Point Reyes bird's beak (*Cordylanthus maritimus* ssp. *palustris*)
2. Marin knotweed (*Polygonum marinense*)
3. Fragrant fritillary (*Fritillaria liliaceae*)
4. San Francisco owl's clover (*Triphysaria floribunda*)
5. Marin manzanita (*Arctostaphylos virgata*)
6. Mount Vision ceanothus (*Ceanothus gloriosus* var. *porrectus*)

Of the ten sites monitored, only one site appeared to have been adversely impacted by the Vision Fire. A San Francisco owl's clover population, located on a ranch road, yielded only two individuals in 1996, a much lower count than the 474 individuals surveyed at the same site in 1988. Dozers disturbed the ranch road at this location during the fire (Semenoff-Irving 1996).

Revegetation

In the entire burned area, only one ¼-acre site required active revegetation. Locally collected coyote brush (*Baccharis pilularis*) cuttings were propagated at Point Reyes nursery in 1996 and were planted along a severely eroded road cut (which was recontoured during post fire rehabilitation) in Muddy Hollow in spring of 1997. Cuttings from new growth on burned stumps had a higher survival rate in the nursery than cuttings taken from shrubs which had not burned at all. Only a few plants survived the summer in the ground, but by fall, red alder (*Alnus rubra*) seedlings had become established.

Photodocumentation

Immediately following the fire, 29 permanent photopoints were established in sites of varying burn intensity. Yearly photographs taken at each site during the peak of the growing season illustrate vegetation recovery over time.

During the second year of vegetation monitoring, a pronounced difference in density and plant height was observed among populations of Australian fireweed in bishop pine forest. Populations of fireweed that had been removed in 1996 appeared significantly less dense in 1997 than populations that had not been treated. May 1997 photographs taken of treated and untreated sites reveal pine seedlings up to two decimeters taller where fireweed had been removed. Documentation of this type of competition is useful for its visual impact as an example of a species' invasiveness.

Recommendations

This section offers recommendations for future post-fire monitoring programs based on experience gained from the Vision Fire. The recommendations have been divided into three categories: pre-fire, during fire and post-fire.

Pre-fire:

- It is crucial to acquire reliable GPS units and have staff trained to use them. If possible, all rare plant and noxious weed locations, trails, roads and landmarks should be GPS'ed and digitally mapped.

- Learn as much as possible about the biology, life history, and response to fire of the weeds and rare plants in your park. Knowledge of aspects of weed ecology (e.g., the growth stage that will exhibit the greatest mortality when treated, seed longevity) will significantly increase your effectiveness at weed control. In addition, knowledge of the most effective control methods for your weed species will increase the success and cost-effectiveness of post-fire weed control.

During Fire:

- Whenever possible, limit dozer activity. Assign a qualified resource advisor to the fire who knows the local topography and vegetation characteristics. Strongly emphasize ongoing communication between the resource advisor and the Incident Commander or the Planning Section Chief. Get involved in the BAER planning process. Park employees who are directly involved in assisting with resource damage assessments become an invaluable source of information once the BAER team has been demobilized.

- Exercise caution when making decisions about post-fire seeding of burned areas. The pressure to seed may be great. Seeding often is not necessary and is ecologically unsound in many situations. Native, viable seed is present in the soil and seed will be dispersed into the burned area from adjacent unburned sites. The native seed will have a better chance of germinating if it is not competing with quantities of non-local seed.

- Take slide photographs of all activities. These will be valuable for historic records and public education. Label and archive slides as soon as possible.

Post-fire:

- Digitally map fire boundaries and all dozer and hand lines immediately. Use this technology to produce maps suitable for field monitoring.

- Hire monitors and organize weed removal crews early in the growing season. Assess equipment and tool needs and place orders immediately. Be prepared to adapt project priorities and crew size as the season progresses.

- Establish a database for managing weed removal data. These data are often crucial for funding requests and public information.

- Develop and implement quantitative, statistically valid protocols for all post-fire vegetation recovery (general vegetation, rare plants, weeds). Complement the quantitative monitoring with qualitative assessments such as photodocumentation. Establish permanent vegetation recovery and erosion photopoints. Ensure funding is secured to continue monitoring for at least three growing seasons.

- Direct public concern toward participation in volunteer restoration programs. Consult with other parks, they often have successful programs and ideas to share.

- Collect seed and cuttings where appropriate for possible revegetation needs. Consider seeding or revegetating with natives at sites of large-scale weed removal. Clearing a site of one non-native species may simply enable a different non-native to encroach. Establish control plots inside and outside the burn to measure the effectiveness of weed control methods. Take “before and after” photographs of weed removal sites.

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Fire Management in the Saguaro Shrub

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Abstract

Beginning in 1992, the USDA Forest Service's Pacific Southwest Research Station, the California State University, San Bernardino, and the Tonto National Forest (TNF), Arizona, initiated an administrative study on fire and fuels in the saguaro shrub community in the Sonoran desert of southwestern Arizona. Study results provided the TNF staff with scientific evidence that enabled them to resolve some conflicts between the conservation of natural resources and user demands. After scientists documented the catastrophic effects of wildfires in the saguaro shrub, resource managers were able to convince administrators to modify fire management policy to include enhanced fire protection with stronger fire suppression measures. The improved fire management program is designed to decrease ignitions and limit fire size by regulating human traffic, reducing fuels in heavy use areas, and expanding the fire suppression program.

Introduction

Resource managers are responsible for ensuring the long-term ecological health of the saguaro shrub vegetation on the Sonoran desert lands of the Tonto National Forest (TNF) in southwestern Arizona. However, programs addressing multiple use and fire management in desert ecosystems are difficult to develop (Wright 1988). Giant saguaro cacti (*Carneigea gigantea*), scenic desert vistas, and old west legends lure an increasing number of visitors to the TNF each year (Figure 1). Recent increases in urban population, eco-tourism, and evolving recreational use have shifted the emphasis of resource use on the TNF from range and commodities to recreation and conservation. These changes have created new fire management and habitat conservation concerns for the TNF staff.

Previously, grazing and range improvement programs reduced fuel accumulation in the saguaro shrub community. Presently, introduced herbaceous species grow in otherwise barren inter-shrub spaces forming large tracts of contiguous fuels highly conducive to ignition and increased fire spread (McLaughlin and Bowers 1982, Rogers 1986). Interestingly, during the last three decades, the number and size of desert fires has increased on the TNF. During this period, over one-third of the saguaro shrub habitat has burned (Narog et al. 1995, Wilson et al. 1998). Currently, anthropogenic ignitions account for the more problematic wildfires occurring in the highly valued saguaro shrub areas of the TNF.

Figure 1. Giant saguaro cactus (*Carneigea gigantea*) among associated vegetation that serves as fuel for desert fires on the Tonto National Forest, Arizona.

As recently as 1995, wildfires burned thousands of hectares of fire intolerant saguaro and its associated vegetation. Charred and moribund saguaros in a fire-scorched landscape do not retain the desired tourist appeal or habitat quality of unburned saguaro shrub. Additionally, saguaro and its associated vegetation may not regenerate after wildfire. Major conflagrations have converted many areas once populated with saguaro shrub into a less desirable grass-dominated community (Cave and Patton 1984, Narog et al. 1999). Furthermore, after an area has burned, grasses and forbs may invade in even greater numbers, may inhibit re-colonization by native species, and may supply even more flashy fuels for future ignitions (Patten 1978, Cave and Patten 1984). Hence, a cycle of potentially larger and more frequent fires may develop. Consequently, new scientifically based fire management policies need to be developed to reduce further loss of saguaro shrub habitat.

An extensive literature review revealed a tremendous amount of information about the autecology of the saguaro and its keystone status with associated plants and animals (McGregor et al., 1962, Steenberg and Lowe 1976, 1977, 1983; Darling 1989; Pierson and Turner 1998). Research examining fire management and fire effects in this community is found throughout the literature, but few authors addressed the saguaro shrub vegetation as available fuel (Humphrey 1963, Cable 1967, Komerck 1969, Rogers and Steele 1980, McLaughlin and Bowers 1982, Patten and Cave 1984, Rogers 1985, Wright 1988, Thomas 1991, Thomas and Goodson 1992, Robinett 1995). Information on the natural or historic fire-return intervals lends valuable insight into the fire tolerance or intolerance of a plant community. Unfortunately, few records had been made of historic fires in the saguaro shrub, and the nature of the vegetation has left little evidence of fire, unlike the dendrochronological records in adjacent higher elevation forests (Baisan and Swetnam 1990). Because saguaros may take 30 years or more to bloom, have low germination and seedling success, and commonly live 200 years, long-term documentation spanning several centuries may be needed to evaluate how fire-return intervals affect this plant community (Bahre 1985, Schmid and Rogers 1988, Thomas 1991). However, waiting two centuries to evaluate fire effects in the saguaro shrub on the TNF was not an option.

In the early 1990s, fire policy on the TNF emphasized a fire prevention program, not fire suppression. Controversy arose from contradictory anecdotal reports of saguaro shrub surviving fire or conversely, saguaro shrub being converted to grassland. Burdened with limited fire management funds, administrators advocated a fire prevention approach and allocated funds for use in activities that would reduce the chance of fire. Because desert land use was evolving, the fire management program, policy, and operational practices needed to be updated. Resource managers touted increases in fire suppression resources such as more more fire crews and engines to solve the fire problem. As a result, documentation and quantification of fire effects in the saguaro shrub were required before fire management protocol would be changed.

To improve fire management policies and programs, the TNF allocated funds for an administrative study on fire in the saguaro shrub habitat. This paper discusses the collaborative research effort conducted by scientists from the Pacific Southwest Research Station, USDA Forest Service, California State University at San Bernardino and the TNF that examined fire and fuels in the saguaro shrub community. Before implementing the study, scientists conferred with TNF resource managers about their concerns. The comments raised included:

- Fire in the saguaro shrub;
- The benefits of grazing for fuel reduction versus its detrimental affects on saguaro regeneration;
- That desert fuels with respect to type, proximity, amount, and introduced species, caused the most serious fire problems;
- Recent natural lightning strike fires were extinguished by the accompanying rain and did not generally become major conflagrations; and
- Anthropogenic conflagrations were becoming a serious threat to the remaining saguaro shrub.

The conclusion was that scientifically based information was urgently needed to expedite the decision making process before more saguaro shrub were lost to wildfire.

Methods

In 1992, scientists surveyed recently burned areas on the TNF to establish study sites. In May 1993, a series of arson fires burned about 1,200 ha (3,000 acres) of a designated “scenic vista” of saguaro shrub on ‘The Rolls”, near Four Peaks Road, Mesa Ranger District, TNF (Figure 2). The Mesa Ranger District staff believed that saguaro and associated vegetation would be completely lost at this Vista View Fire site and that the habitat would be converted to grassland--a phenomenon that they had observed locally in other burned areas. They were anxious for quantitative saguaro mortality data for verification of fire effects for management reports. This area had burned and unburned vegetation in close proximity and was chosen as our study site.

The composition and structure of the saguaro shrub community was described from a fire and fuels perspective on seven 1 ha plots and six 350 m point quarter plots. This included specific information on vegetation abundance, distribution patterns, fire injury, and mortality. Field crews composed of research and Mesa District personnel collected densities, frequencies, and dimensions on the burned and unburned vegetation in the area of the arson set fires.

In 1995, another large wildfire occurred on “The Rolls” on the Mesa Ranger District. This summer wind-driven River Fire burned more than 4,000 ha (10,000 acres) of saguaro shrub (Figure 2). Two previously established unburned study plots were burned in the River Fire. We modified our experimental design and incorporated this second fire into our study. Some speculated that this huge wildfire could have been limited to about 200 ha (500 acres) if fire fighters had not lost the use of an air tanker for water dropping.

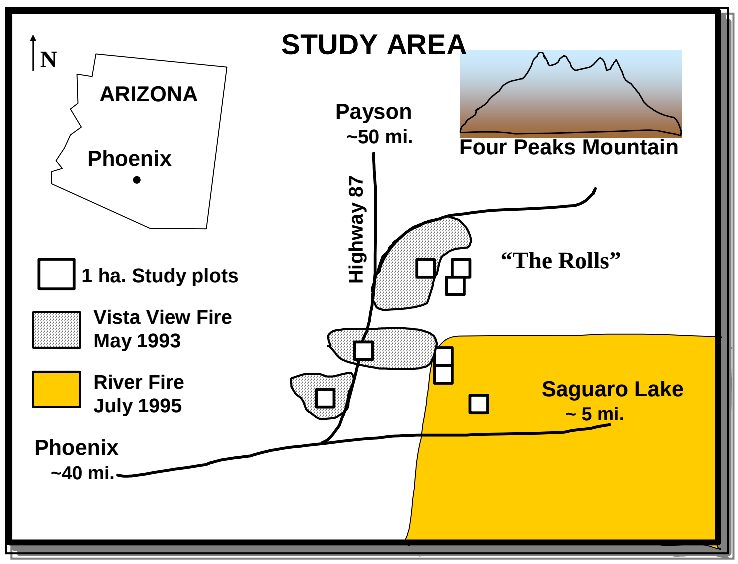

Figure 2. Data was collected in unburned and burned saguaro shrub on the Mesa Ranger District of the Tonto National Fires in southwestern Arizona. (Map not to scale)

Results

Data from the 1.5 year post-burn survey of the Vista View burn study documented that 19% of the saguaros had died. Of the saguaros that were alive, 90% had sustained some fire injury; however, some of the fire injury appeared nominal, suggesting that not all saguaros on this burned site would die (Wilson et al. 1996). Many seriously burned saguaros still maintained photosynthetically active tissue and bloomed after sustaining fire injury. Apparent thermal damage measured on sampled saguaros was used to make preliminary predictions about their long-term survival. Because of delayed mortality, 5 or more years may be required before absolute mortality determinations can be made (Rogers 1985). After documenting the initial survival and injury of the burned saguaro, we estimated the potential mortality (Narog et al. 1999; Wilson et al. 1995a, 1995b, 1996, 1998).

The Vista View Fire provided insight into the post-fire growth response of the saguaro’s associated vegetation. Reestablishment of the saguaro’s associated vegetation is necessary to provide shelter as nurse plants for future saguaro generations (Vandermeer 1980). Preliminary findings reported 29 out of 30 plant species were resprouting after the burn. Of these, six woody plant species comprised 88% of the nearest neighbors to the saguaro. Nearest neighbors that serve as nurse plants may also provide fuel for fire. The arrangement of these fuels occurred in dense patches showing a 25% overlap. Between these patchy fuels were varying degrees of herbaceous material (Wilson et al. 1998). On the basis of estimates of the burned skeletons that framed new growth, many plants had recovered about one-third of their preburn size in less than 2 years after the fire. This was important information for implementing recovery programs because resource managers did not realize that many of the desert plant species were able to resprout (Wilson et al.1995a).

The two wildfires were different in fire behavior and fire effects on the vegetation. Many saguaros sustained thermal injury in the Vista View Fire, but the River Fire reduced many saguaros to woody skeletons or less (Figure 3). Initial saguaro mortality was 5% greater on the River Fire than on the Vista View Fire, and long term mortality may be even greater. Also, site regeneration after the River Fire may be slower because only 21% of the associated vegetation resprouted compared with 94% in the Vista View Fire.

Annual reports of preliminary research findings have been made to TNF staff to facilitate transfer of information. 'Unseasonable' burning from anthropogenic ignitions was documented to be harmful to saguaro populations and the associated vegetation. Further documentation of the saguaro shrub community is needed to determine the long-term impact of these fires. During the interim, land valuation of the saguaro shrub resource may justify greater fire resource expenditures.

Figure 3. Fire impact on saguaro shrub habitat--4,049 ha River Fire, July 1995, Tonto National Forest, Arizona.

Discussion

Because no single factor contributed solely to the fire risk, fire hazard in the saguaro shrub must be addressed from more than one approach. The interactions among research scientists, resource managers, and policy makers facilitated the development of modified and improved fire management policies for this vegetation. Our data generated new information about distribution of fuels and post-fire effects in saguaro shrub. Saguaro shrub survival and mortality varies among fires and substantiating predictions of final post-fire outcomes requires long-term observations. However, the extreme nature of the River Fire brought suppression and prevention advocates together and emphasized the need for both fire management approaches. This signified a shift in perspectives from a predominantly fire prevention program to an integrated or balanced program that included critical fire suppression resources and continued research support. Concurrently, media coverage of the 1995 fire~~s~~, “the Rolls”, brought public attention to the little recognized fact that fire had become a serious threat to the saguaro shrub in Arizona’s Sonoran desert. Political and public pressure and support to remedy the fire problems in this precious desert ensued. Because large areas of saguaro shrub has already burned on the TNF, the risk of yet another major conflagration is unacceptable at this time.

A non-traditional economic valuation of environmental resources in the saguaro shrub was needed to justify financial and resource backing to fight fire. To establish an economic value for the desert land, scientists recommended converting the number of saguaros per unit area to dollars. Data collected during research on “The Rolls” determined that each hectare of land supported about 20 mature saguaro. In the mid-1990s, mature saguaro sold at market prices between $5,000 and $10,000 each. Assuming a conservative cost of $5,000 per saguaro, an estimated value for 1 hectare (2.47 acres/ha) of saguaro shrub land would be valued at $100,000 (@ $40,000/acre). This estimate is based solely on the value of saguaros on that parcel of land. For example, the River Fire ravaged more than 4,000 ha (10,000 acres) of prime saguaro shrub land. Assuming all saguaros on this burned land die, we can estimate an economic loss of nearly $400 million! If only 25% of the saguaro are killed by this wildfire the calculated loss in saguaro alone would still be substantial--$100 million. These computations easily convert the ‘low value’ of scenic desert land into a very ‘costly and desirable’ commodity. Furthermore, these calculations do not include the high cost of fire fighting resources or dollars lost due to decreased visitor use. This economic valuation of the saguaro shrub helped the TNF staff to justify additional fire resource expenditures.

New scientific information, economic justification, and public and political concerns were combined to support a change in policy from fire prevention to an integrated fire program. Fire suppression resources were increased so that future fires in the saguaro shrub would be immediately extinguished. Expanding the fire suppression program was necessary because reducing the fire hazard created by the flammable introduced herbaceous plants would be a formidable task. An integrated program combining both enhanced fire prevention and fire suppression was feasible if ignition points rather than available fuels were the focus. Fire prevention efforts could be directed at the most logical ignition points: high human use areas. Enhanced fire prevention was futher accomplished by measures such as closing roads, redirecting visitor activities, reducing fuels, not seeding in high use areas and educating the users about fire hazards. Fire safety campaigns are necessary to educate visitors to the fact that ‘deserts burn’ and valuable resources can be lost. For example, the TNF could prohibit the use of products known to contribute to ignitions, such as the type of bullet that ignited the River Fire as a result of its high spark potential. If high fire hazard predictions are made, then the TNF staff should adjust resource use so that the saguaro shrub vegetation can be adequately monitored and so that additional fire resources are available to address conflagration emergencies.

Conclusion

Preliminary research findings are already helping TNF resource managers explore new perspectives, balance conflicting needs, and justify their decisions when applying newly modifed resource management policies. The improved fire management program on the TNF is aimed at preventing ignitions by regulating human traffic and reducing fuels in heavy use areas and implementing an active suppression program to limit fire size. After two consecutive El Niño years of high precipitation and a corresponding increase in herbaceous fuels, the Mesa Ranger District resource managers were able to report that the saguaro shrub habitat had a reduction in ignitions and acres burned for the 1998 fire season. Our cooperative research with the TNF investigating fire concerns in the saguaro shrub provides a positive model that improved fire management policy. With public and political support, scientists, resource managers, and policy makers worked together for the benefit of the ecosystem. Developing a better understanding of fire ecology in the saguaro shrub community helped to reduce fire risks and costs.

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Research and Restoration of Dynamic Ecosystem Processes: The Warner Fire Process Research Natural Area Proposal

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Introduction

In October, 1991, arsonists ignited the Warner Creek Fire inside a protected Habitat Conservation Area managed for the Northern Spotted Owl (Strix occidentalis caurina). The wildfire burned across 8,900 acres, including the entire 6,800 acre Cornpatch Inventoried Roadless Area, and became the second largest, costliest wildfire in the history of the Willamette National Forest. Fearing that the fire-killed snags and logs posed a risk of "catastrophic" reburn, the Forest Service proposed extensive salvage logging in order to reduce fuel loads and construct fuel breaks. This sparked a firestorm of controversy among conservationists who feared copycat arson-for-salvage incidents would occur in other protected habitat reserves. Over the next four years, the Warner Salvage Sale became one of the most controversial, contested timber sales in the country. The Clinton Administration finally withdrew the timber sale in 1996, and no salvage logging ever occurred inside the wildfire area. Currently, the Warner Burn represents one of the rarest landscapes in the Cascadia bioregion: a roadless, mid-elevation, largely unmanaged burned forest containing both young natural stands and high-mortality old-growth stands.

During the conflict over the Warner Salvage Sale, a collaboration among scientists, conservationists, and Forest Service employees proposed that the Warner Burn be managed as a Fire Ecology Research Natural Area (RNA) to research and restore natural fire disturbance and succession processes. This idea was first articulated in "Alternative EF: Ecology of Fire" published in the Warner Fire Recovery Project's Final Environmental Impact Statement. Dubbed the "Know Action Alternative" drafted by "citizen-scientists," Alternative EF set a number of precedents for public participation in a fire recovery project. More significantly, it inspired a formal revision of the Oregon Natural Heritage Plan that created a new paradigm of RNAs devoted to dynamic ecosystem disturbance processes such as fire.

The vision of the Warner Fire Ecology RNA has expanded into a more recent proposal that utilizes conservation biology principles to link the Warner Burn with five adjacent Inventoried Roadless Areas and two designated Wilderness Areas. This would form a landscape-scale fire ecology research complex suited to the fire regime of the westside Cascades. Anticipating future progressive developments in fire management philosophy and policy, proponents seek to develop a model fire management plan for the RNA that would facilitate research and restoration-oriented prescribed burning, wildland fire monitoring, and innovative minimum-impact suppression techniques. However, as the Warner RNA Proposal evolves, it encounters increasingly complex social, scientific, and management issues that challenge not only our desires to "learn from the burn," but also our abilities to live with wildland fire.

The Warner Creek Fire

Ignited by arsonists during extreme drought conditions, the Warner Creek Fire entered the old-growth tree crowns almost from the point of ignition, and surged rapidly upslope. During a blow-up event, nearly 3,000 acres of prime spotted owl habitat was severely burned in a single afternoon. Stands of old-growth douglas-firs, western hemlock, and western red cedar were completely scorched from ground to crown. In other areas, the fire was a low-intensity underburn, causing little or no mortality on the overstory trees. Within the core of the Warner Burn nearly 500 acres went completely unburned as wildfire failed to ignite an ancient forest grove in the deep, dark Kelsey Creek basin. Forest Service scientists later determined that the Kelsey Creek old-growth grove for at least 850 years had not burned.

Over 2,500 firefighters were dispatched to battle the blaze, supported by two fire camps and an armada of equipment, vehicles, and aircraft. Resource advisors from the local Oakridge Ranger District were assigned to the fire, and they helped provide guidance to firefighters on minimum impact suppression tactics in owl habitat stands. Nevertheless, some "heavy-handed" impacts did inevitably occur as a result of aggressive suppression incidents. For example, chemical retardants were dumped into streams, numerous trees were felled along a popular hiking trail, a mile-long dozerline was plowed deep inside the Roadless Area, and thousands of acres of owl habitat were burned in huge backfires and burnout actions, accounting for an estimated 30% of the total burned acreage. The environmental impacts and economic costs of the $10 million suppression effort were the initial underlying motivations for the citizens to create an RNA alternative. The RNA proponents wanted to protect the wild "owl-growth" forests in the Warner Burn from future aggressive fire fighting as well as from commercial salvage logging.

Even though the Warner Creek Fire was started and spread by unnatural ignition sources—arsonists and firefighters--the resulting mosaic of fire effects was representative of the natural fire regime of the westside Oregon Cascades. The fire was especially effective in restoring the natural ridgetop meadow complex of the aptly-named Cornpatch Inventoried Roadless Area (in the autumn, the meadows turn the color of maize). These meadows had historically been maintained by frequent lightning ignitions and Native American burning, but were now shrinking in size and species diversity due to fir encroachment from fire exclusion. In this and several other respects, the ecological effects of the Warner Fire contradicted the "catastrophic wildfire" depiction that was made by Willamette National Forest representatives.

The Warner Fire Recovery Project

Two weeks after the Warner Creek Fire was controlled, the Forest Service initiated the Warner Fire Recovery Project. The stated purpose and need for the Project was to recover spotted owl habitat and increase knowledge about owl habitat recovery. The secondary underlying need—to increase knowledge—later proved vital in legitimizing an RNA alternative. The Warner IDT originally assumed that the Interagency Scientific Committee's standards and guidelines for HCAs would prohibit salvage of any downed or standing trees (Thomas, et al 1990). Therefore, the IDT determined that the Project provided "an opportunity to set aside all or a portion of the fire area for studies of how both natural and managed landscapes respond to large scale fires" (USDA-FS 1992) Research Opportunities thus became a significant issue for analysis in the EIS. The Project's Decisionmaker (Willamette FS Supervisor, Darrel Kenops) instructed the IDT to include a research alternative in the Draft EIS. Unfortunately, the Decisionmaker's idea for a research alternative was aimed solely at monitoring the effects of managed recovery actions--primarily salvage logging and conifer planting--on owl habitat recovery.

The Decisionmaker rationalized the need for salvage logging by defining owl habitat recovery in terms of wildfire protection, and claimed that salvage logging was a tool that would "increase our ability to fight future fires...and reduce the likelihood of fire damaging developing owl habitat" (USDA-FS 1992). However, the IDT's initial idea of setting aside the Warner Burn for the study of natural fire recovery processes was quickly adopted by members of a special citizen advisory group organized by the Willamette to give regular input on the Recovery Project. A majority of the Warner Public Participation Group formally requested that a new alternative be created that would designate the Burn as an RNA, and develop a new fire management plan utilizing Prescribed Natural Fires. Regardless, the Decisionmaker declared that both RNAs and fire management planning were issues "outside the scope of the project," and excluded these from the Draft EIS.

In response, members of the citizen advisory group networked with fire scientists and forest ecologists from across the Pacific Northwest to develop their own proposal for an RNA-based fire recovery plan, and submitted this to the Project Decisionmaker during the public comment period. In accordance with the agency's nomenclature for its range of alternatives, the proposal was called "Alternative EF: Ecology of Fire." Later, when the Willamette's Draft Preferred Alternative was withdrawn by the Forest Service's Owl Oversight Committee, who determined that salvage logging 40 million board feet from 1,200 acres of the Roadless Area portion of the HCA was inconsistent with the interagency owl conservation strategy, the Decisionmaker allowed Alternative Ecology of Fire (Alt. EF) to be fully developed, analyzed, and published in the Final EIS. This set in motion a truly collaborative effort amongst agency specialists, conservationists, academic and independent researchers that continues to this day, further developing and refining the concept of a fire process RNA for the Warner Burn.

Alternative EF: Ecology of Fire

The goal of the authors of Alternative EF (Alt. EF) was to propose a recovery strategy that centered around the restoration and research of wildland fire processes. This would offer a genuine *alternative* to the status quo "alternatives" that proposed salvage logging and continued fire exclusion. Alt. EF proposed a fire management strategy using Prescribed Natural Fires (PNFs) to both recover the resource (owl habitat) and protect the site from a future severe wildfire. The entire Burn was divided into four zones each distinguished by priority-valued natural resources and PNF prescriptions that ranged from low to moderate and high intensity. Thus, for example, the zone containing spotted owl nest sites was assigned a low-intensity prescription, while the ridgetop meadow zone was allowed a high-intensity prescription. Along with PNFs, an active program of fire monitoring and fire effects research was proposed in order to study both natural fire disturbances and recovery processes.

Alt. EF prioritized PNFs, but also allowed some management-ignited prescribed fires, especially in the ridgetop meadow zone. Such fires could occur only after intensive fire history research that included anthropological research on the frequency, locations, and methods of historic Native American burning along Bunchgrass Ridge. In a move that preceded the Federal Wildland Fire Management Policy and its concept of "Appropriate Management Response," Alt. EF allowed limited suppression activities to occur in order to keep natural fires within their prescription. This dissolved the strict dichotomy that dictated fires be managed either as PNFs with no suppression activities, or be declared wildfires and receive total suppression. In the event that some fires exceeded their prescribed fire intensity, Alt. EF mandated that only light-hand/minimal-impact suppression could be used within the Burn. The concern was that aggressive suppression actions would not only damage the environment, but might destroy ongoing research projects and monitoring plots.

The Decisionmaker and IDT believed that both prescribed and wildland fires would adversely affect owl nesting and roosting habitat. However, unbeknownst to the authors of Alt. EF at that time, the fire and fuels management team for the "CASPO" Report urged that prescribed underburning be used to prevent stand-replacing wildfires in California spotted owl Protected Activity Centers (Vener et al 1992). In response to critics who claimed that Alt. EF was too "radical" because it promoted wildland fire use instead of fire exclusion in owl habitat reserves, the proponents of Alt. EF countered that its reliance on natural processes was actually quite "conservative." Proponents argued that, logically, spotted owls evolved with recurring fires and natural succession, and observed that most existing owl stands exhibit some evidence of past fires. Proponents added that fires were a prime agent creating multi-storied canopies, large snags and logs that were vital components of superior spotted owl habitat. Furthermore, proponents took issue with the agency's strategy of protecting owl reserves with fire exclusion─a policy that had already been "vetoed" by the arsonists who ignited the Warner Creek Fire. Finally, proponents took comfort in the fact that the resident population of spotted owls continued to inhabit and successfully reproduce in the Burn. Indeed, the continued existence of the owls in the Burn challenged the agency's assumption that it had to do any managed recovery actions at all, especially salvage logging to increase fire suppression effectiveness.

Alt. EF was endorsed by prestigious academic members of the research community, including some of the scientists who participated in FEMAT and helped design the Northwest Forest Plan. The student governments of Oregon's two largest universities passed official resolutions in support of Alt. EF and sent these to Forest Service Chief, Jack Ward Thomas. Then, the Forest Service's Regional RNA Coordinator and ecologists from the Pacific Northwest Research Station drafted their own RNA proposal for the Warner Burn. Finally, the Oregon Natural Heritage Advisory Board used the Warner Creek Fire and Alt. EF as inspiration to establish a new "Fire Process" cell for their network of RNAs. This created a qualitatively new kind of RNA aimed to protect areas for their dynamic ecosystem processes rather than static species composition or geologic features. Consequently, in the Willamette's final recovery plan the Decisionmaker set aside a 4,200 acre portion of the Burn as a "Natural Succession Area" (NSA) to be later considered for designation as an RNA. Unfortunately, the NSA was also going to be bordered by extensive salvage clearcut units, and fragmented into six sections by fuelbreaks.

Implementation of the Willamette's final recovery plan was repeatedly delayed, first by a dozen administrative appeals, and then a lawsuit pressed by local conservationists. All these delays occurred during a time of rapid change in forest and fire management policies initiated by the Northwest Forest Plan and the Federal Wildland Fire Management Policy and Program Review. Finally, in the midst of a nationwide protest campaign whose slogan was "Stop the Warner Salvage Sale; Save the RNA!" President Clinton ordered the Warner salvage sale to be withdrawn in August, 1996, and no salvage logging ever occurred inside the Warner Burn. The Willamette then essentially abandoned the fire recovery project, choosing the No Action Alternative by default. Into this management void came a new, more expansive RNA proposal for the Warner Burn.

The Warner Fire Process RNA Proposal

With the fire recovery project nullified, the group of self-named "citizen-scientists" who had drafted Alt. EF developed a new RNA proposal, and submitted it to the Regional RNA coordinator in September, 1997. Called the "Warner Proposal" for the sake of brevity, it was not confined to the 9,000 acre wildfire perimeter. New boundaries were drawn along suitable topographic features and landforms that would aid fire confinement strategies and minimize the need for aggressive suppression. Using conservation biology principles, the Warner Proposal linked the Burn with a cluster of four other Inventoried Roadless Areas (RAs). These RAs, in turn, were located adjacent to two contiguous Wilderness Areas along the Cascade Crest that have recently approved PNF plans. The resulting vision: a 48,000 acre RNA connected to 336,000 acres of Wilderness, where it is hoped that large-scale wildland fires could be managed for restoration and research purposes.

The Willamette responded to the Warner Proposal by resurrecting the Decisionmaker's 4,400 acre NSA-RNA proposal—minus the fuelbreaks and clearcuts--and submitted it to a public scoping period. Over 1,000 public comment letters poured in, and much to the agency's shock and dismay, every single letter rejected the Willamette's proposal! The chief criticism of the research and conservationist communities was that the NSA-RNA was too small and its boundaries too arbitrarily-drawn to allow wildland fire use. Instead, the letters demanded that the citizen-scientists' Warner Proposal be included in the NEPA process. This prompted the Willamette and the Pacific Northwest Research Station to host a roundtable workshop of fire scientists from Oregon, Washington, and British Columbia to discuss design and management criteria for Fire Process RNAs. The assembled scientists validated the principles articulated in both Alt. EF and the new Warner Proposal, agreeing that landscape-scale RNAs were needed in order to best capture natural fire processes at the spatial and temporal scales they function in the westside Oregon Cascades. Confronted by the scientists' implicit endorsement of the Warner Proposal, and lacking any public support for its own NSA-RNA idea, the Willamette has quietly put the NEPA process on hold.

Ongoing Activities to Learn From the Burn

Despite the lack of formal RNA status, the Warner Burn has attracted fire ecology research and educational activities throughout the last eight years. For example, 45 monitoring plots have been established by Forest Service ecologists in order to document the early structure, composition, and regeneration of the Burn. If future Forest Service budgets allow, these plots will be resurveyed as part of a long-term monitoring plan. Scientists from the University of Oregon State and Oregon State University have also conducted some studies on soils, vegetation, and wildlife in the Burn. The Cascadia Fire Ecology Education Project (CFEEP), a nonprofit conservation organization, and the Northwest Youth Corps (NYC), an alternative high school for at-risk youth interested in pursuing forestry careers, have embarked on a partnership to establish long-term fire monitoring plots. CFEEP and NYC have initiated a unique snag longevity study to monitor the rate of fall and decay of fire-killed snags and logs over the next several decades. The methodology for the snag study was developed with the help of Forest Service ecologists and sustainable forestry consultant, Chris Maser (Masser 1988). The nonprofit groups intend to begin applying the National Park Services' fire effects monitoring protocol to their current and future research plots in the Burn.

Additionally, students from all across the country have attended special fire ecology field seminars and research outings in the Warner Burn as part of curriculum for the Wildland Studies Program and the Cascade Science School. The Warner Burn is conveniently located next to a major highway just one hour's drive from the city of Eugene, Oregon. Educators from local universities and school districts are becoming some of the more vocal advocates of the Warner Proposal because it has great potential as an outdoor education site and living laboratory for academic research projects. This fulfills one of the oft-neglected purposes of RNAs to foster education, as well as conduct research and conserve biodiversity (USDA-FS 1997). Accordingly, it is hoped that formal protective status as an RNA would enable future generations of scientists and students opportunities to "learn from the Burn."

Controversies and Challenges of Fire Process RNAs

RNA proponents are urging the Willamette to continue the collaboration that created Alt. EF, to proceed with the NEPA process, and to welcome another citizen-scientist Alternative in the EIS. This effort conforms with the Committee of Scientists' Report that encourages up-front collaboration with scientists in Forest Service planning and projects (Committee of Scientists 1999). The Federal Wildland Fire Management Policy and Program Review has also lauded the role of communication and collaboration, and has called for a renewed emphasis on public participation and partnerships in all aspects of wildland fire management (USDA/USDI 1995). If and when the Willamette moves forward with the NEPA process, the public, scientists, resource specialists, and regulators will all want to review and submit comments because the EIS will likely raise several critical social, scientific, and management controversies that are literally "burning issues of our time."

Socioeconomic Issues

The idea that a Fire Process RNA will protect a relatively large area of public wildlands *for* fire and not *from* fire fundamentally conflicts with the Forest Service's dominant paradigm of fire protection predicated on fire exclusion and aggressive suppression. Forest Service managers will have to overcome the Smokey Bear legacy that conditioned the public to generally fear and loath forest fires. The new Federal Fire Policy offers direction for this management challenge: "Agencies and the public must change their expectation that all wildfires can be controlled or suppressed" (USDA/USDI 1995). Consequently, the RNA EIS must be part and parcel of a major fire ecology education campaign to help change attitudes toward wildland fire.

Relatedly, the Forest Service will have to re-educate the public on forest health and salvage logging issues, and the change its common portrayal of burned, fallen, diseased, dead, or rotting trees as a "wasted resource" and/or "wildfire hazard." An RNA designation would preclude intensive management and commodity extraction activities; thus to remove certain acreage containing commercially-valuable timber and set them aside for the "wrath of wildfire" will present a public relations challenge, to say the least. The specific Warner RNA proposal largely mitigates this issue by incorporating lands such as LSRs and RAs that make future commercial timber extraction problematic, and even more of an unlikely scenario if President Clinton's new Roadless Area Initiative results in protection of RAs from further logging and road-building. Indeed, the only commercial logging that would be affected by the 48,000 acre Warner Proposal is the "Helldun" timber sale, a planned clearcut of 100 acres of ancient forest, sanctioned by the Salvage Rider, and strategically located a short distance *outside* of the Warner Burn. The agency will have to educate the public on the many positive ecosystem benefits of wild forests containing burned trees.

The Forest Service will also have to attempt to articulate the many potential socioeconomic benefits that could accrue from a large-scale RNA. Too often, RNAs are considered merely as land "set-asides" or "locked-ups" that only benefit researchers. For example, the IDT for the Warner Fire Recovery Project estimated that salvage logging would have created a total of 90 timber jobs, but could only conceive of just one (1) job in an RNA! The authors of Alt. EF, however, predicted the need for large numbers of public and private sector jobs to plan, administer, and use a fire process RNA. Workers would be needed to establish research plots, conduct field surveys, collect baseline data, initiate research projects, construct trails, monitor wildland fires, and manage prescribed fires. Of course, these activities would have to be funded by alternative sources such as federal appropriations, grants, endowments, etc. rather than traditional commodity-producing projects. But given a suitable funding source, the number and kinds of research/restoration jobs and other socioeconomic benefits that could accrue from a Fire Process RNA over the next century or more is limited only by our imagination.

Scientific Controversies

There are genuine unresolved questions and controversies over the relationship of spotted owls, "wet-side" old-growth forests, and forest fires. These unknowns became the very impetus and driving rationale for the RNA alternative in the Warner Fire Recovery Project. It is probably a safe assumption that a Fire Process RNA will entail some degree of extra risk of potential loss of some components of owl habitat. However, it is hypothesized that given sufficient time, natural succession processes can recover sites from the effects of fire disturbances. Forest Service managers typically prefer to skip early successional stages following fire or logging disturbances; yet, it remains a mystery whether or not fire disturbances and early successional stages are necessary elements of long-term old-growth development. For example, recurring fires could stimulate pulsed regeneration and pruning mechanisms that help develop all-aged, multi-storied canopy structures that are preferred by spotted owls. Likewise, it is recognized that fires are a prime disturbance agent creating large-diameter snags and logs that are vital habitat structures to owls and their prey. An RNA could help reveal how natural fire disturbance and recovery processes function over the long-term, and serve as a control area to compare with managed areas undergoing quasi-experimental silvicultural treatments.

Another controversial issue concerns the relationship of past management impacts inside a new RNA. Presented with the need for large-scale areas to study fire processes, and given the rarity of non-Wilderness lands that have escaped intensive management activities to date, the Oregon Natural Heritage Plan allows up to 10% of an RNA to include sites impacted by past timber management. According to the architects of this plan, however, this provision is a starting point subject to further refinement (Kertis *pers communication*, 1999). In the Warner Proposal, for example, there are approximately 77 miles of logging roads and 8,000 acres of young plantations, which comprise approximately 16% of the landbase within the proposed RNA boundaries. It should be assumed that fire behavior and fire effects will differ in managed versus unmanaged stands, and this may adversely affect the validity of some of the data. Although existing logging roads, clearcuts, and plantations normally would have precluded the establishment of a standard Elemental

RNA, the theory is that a Process RNA allows natural fire disturbance and recovery processes to proceed even in the areas impacted by past management.

Management Conflicts

Other issues to debate in the public forum of the EIS revolve around managing the RNA for "multiple uses" such as restoration, recreation, and public education. For example, prescribed burning could be used as means of restoring and maintaining biodiversity, but could these also be considered suitable "research" activities in an RNA? The frequency of natural ignitions may not be sufficient to restore the natural fire regime of the Warner area given nearly a century of fire exclusion and the elimination of historic Native American burning. Although there are strong advocates for wildland and prescribed fire use in RNAs specifically to maintain ecological processes, most Forest managers have opted for a "hands off" approach to fire management (Johnson 1983). Unfortunately, out of 79 established RNAs that are comprised of fire-dependent plant communities, only five of these RNAs undergo prescribed burning; the rest are declining in species, structural, and seral diversity due to fire exclusion (Greene et al, 1995). Fortunately, one of these five RNAs receiving periodic prescribed burning is located approximately 60 miles from the Warner Burn in an eastside Cascade ecosystem on the Deschutes National Forest. On the Metolius RNA, scheduled prescribed burns are conducted as part of a long-term monitoring program serving both restoration and research objectives (Reigel et al, 1999). The Metolius RNA prescribed burning program offers a working model for Willamette managers to apply to a Fire Process RNA in the Warner Burn.

During the development of Alt. EF, a conflict arose within the research community over the belief by some scientists that recreational and educational activities were inappropriate uses for the RNA. Some scientists opposed the construction of a fire ecology interpretive trail out of fear that off-trail hikers would unwittingly trample upon research plots, or vandals would deliberately destroy research sites and equipment. The Warner Proposal, however, includes sufficient acreage that the potential risk of a catastrophic loss of research plots should be minimized. And protection of research sites should be a prominent theme for interpretive and educational programs. National Parks must frequently grapple with the dilemma of providing for "recreational playgrounds and natural area preservation;" hence, National Park Service employees may want to lend some fraternal advice to the Forest Service on how to manage for these seemingly contradictory uses of the Warner RNA.

Perhaps the most perplexing issue concerns the current general lack of funding for the RNA program in the Pacific Northwest Region (Greene, 1999). The Willamette Supervisor estimates that it would cost approximately $200,000 to conduct a NEPA process, but claims that there is no money in the current budget to publish an EIS, let alone fund fire management planning or research projects for the RNA. However, RNA Proponents claim that if the desire exists, then the Forest Service could graft funding from multiple accounts, regions, and agencies to get the RNA established. Perhaps the Warner RNA could serve some purposes and receive funding from the Joint Fire Sciences Program. Once the bureaucratic paperwork is done, the example of the privately-funded CFEEP-NYC research program demonstrates that the federal government could tap into local community labor resources to get some of the "grunt work" of baseline data collection and field research projects started. Managers are thus being urged not restrict themselves to their agency's internal resources of budgets, staff, and even expertise, but look to forming collaborative partnerships with the research and conservation communities to supplement limited federal resources.

Conclusion

At the time of this writing (January, 2000) the current status and future fate of the Warner Fire Process RNA proposal is uncertain. On the one hand, Forest Service officials now claim that the agency does not have enough money in its current budget to fund an EIS, and have offered no assurances that the NEPA process will move forward. On the other hand, the Warner Proposal continues to generate interest and endorsements from scientists, researchers, educators, and resource specialists in the Forest Service and other land management agencies. Many proponents in the research and conservation communities look beyond the Warner Burn, hoping to establish a network of Fire Process RNAs across the western U.S. In the face of apparent internal organizational resistance to the Warner Proposal, Oregon political representatives such as Governor John Kitzhaber, Senator Ron Wyden, and Congressman Peter DeFazio have written the Willamette expressing their interest in the RNA and urging the agency to complete the NEPA process. Meanwhile, the Warner Burn attracts people as a place of beauty, mystery, and discovery. If and when the visionary Warner Fire Process RNA Proposal ever becomes reality, it will function as part and parcel of emerging paradigms and changing policies of forest and fire management.

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