

Wilderness Fire Science: A State-of-Knowledge Review

James K. Agee

Abstract—Wilderness fire science has progressed since the last major review of the topic, but it was significantly affected by the large fire events of 1988. Strides have been made in both fire behavior and fire effects, and in the issues of scaling, yet much of the progress has not been specifically tied to wilderness areas or funding. Although the management of fire in wilderness has been slow to recover from the fires of 1988, science has progressed most significantly in its ability to deal with fire at a landscape level. Major challenges include better understanding of the regional context and function of wilderness areas, as well as understanding and incorporating fire patchiness, variability and synergistic disturbance factors into predictive models. If more precise models are to be applied accurately in wilderness, better weather databases are essential.

Wilderness fire has presented both managers and scientists with considerable challenges over the 30 years that wilderness fire programs have been operational. Wilderness fire, in its purest form, should be “wild” fire: unfettered by the constraints of humans. We have never prescribed a “let-it-blow” policy for tornadoes and hurricanes, a “let-it-erupt” policy for volcanoes or a “let-it-grind” policy for glaciers. Why, then, did we need a “let-it-burn” policy for fires, or surrogate strategies like prescribed fire? Humans and fire have an inseparable history (Pyne 1995). We have been able to control fire for human purposes for thousands of years and find it very difficult to “let wild fire loose” (Pyne 1989). There are some good reasons for this reluctance, including the issues of safety to humans and damage to resources and property. As much as we have tried, we have not been able to find areas large enough to “let wild fire loose,” and this has been at the root of the challenges to research and management over three decades. It remains a primary challenge today.

The literature on fire in wildlands is immense. As in every field, some of it is hardly worth printing, while some is insightful and informing. In this review, I cannot cover even the entire latter category, and do not attempt a complete literature review by any means. My objective is to summarize the major trends in wilderness fire science since its inception, with a focus on recent times, and to define scientific challenges for the future. Fortunately, there are a number of major conference proceedings that

have synthesized fire research over the past decades and allow somewhat cursory coverage in this review. In chronological order, they include: Fire Regimes and Ecosystem Properties: Proceedings of the Conference (Mooney and others 1981); Proceedings – Symposium and Workshop on Wilderness Fire (Lotan and others 1985); National Wilderness Research Conference: Issues, State-of-Knowledge, Future Directions (Lucas 1986a) and National Wilderness Research Conference: Current Research (Lucas 1986b); Fire and the Environment: Ecological and Cultural Perspectives (Nodvin and Waldrop 1991); and Proceedings: Symposium on Fire in Wilderness and Park Management (Brown and others 1995a). In addition, there is the once-annual and now-periodic Tall Timbers Fire Ecology Conference proceedings which contain significant material related to wilderness areas. Several books are available that provide specific geographic or disciplinary information about fire: Fire and Ecosystems (Kozłowski and Ahlgren 1974); Fire Ecology of the United States and Southern Canada (Wright and Bailey 1982); and Fire Effects on Ecosystems (DeBano and others 1998). Some regional treatments have been possible where information is abundant: Fire and Vegetation Dynamics: Studies from the Boreal Forest (Johnson 1992); and Fire Ecology of Pacific Northwest Forests (Agee 1993).

Definitions of fires have changed over the past decades, most recently in 1997. I have attempted to be faithful to the new terminology where possible, but doing so is awkward. The first natural fires allowed to burn were called “let-burn” fires, but that phrase conjured up an impression of no management at all. It was changed to “prescribed natural fires” in the 1970s as part of a tripartite division of fires: wildfires, which were unwanted fires of any origin, prescribed fires, which were manager-ignited fires, and prescribed natural fires. All of these fires were called wildland fires, as they occurred in wildlands, in contrast to structural fires. In the mid-1990s, Federal fire policy was reviewed, and a new terminology was created. Prescribed fire remained a separate category, and all other fires were classed as “wildland fires,” which was somewhat confusing as that phrase referred previously to all fires in wildlands. The wildland fire category was subdivided into (1) wildfires (unwanted wildland fires) and (2) wildland fires that might be managed (those of natural origin burning within a predetermined zone and within prescription limits of some type): the old prescribed natural fire. Unfortunately, there has been no formal phrase adopted for these fires: Prescribed natural fire is now defined by what it is not (not a prescribed or unwanted wildland fire). A logical name such as “managed wildland fire” is not very descriptive or formally used, so I will continue to call these fires “prescribed natural fires,” or “pnf.”

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Historical Evolution of Fire Science Applied to Wilderness

The recognition of ecological process as a major management objective for parks and wilderness came of age in the 1960s. Before then, of course, there were national parks and monuments managed by the National Park Service (NPS) and designated wild areas and primitive areas, as well as considerable unroaded but unclassified lands, managed by the Forest Service. Fire was suppressed in all of these units, except for experimental burning in Everglades National Park (Robertson 1962). Three major public policy shifts occurred in one decade: the Leopold Report (1963), the Wilderness Act (1964) and Department of the Interior fire policy (1968) that recognized natural processes, including fire, as valid objectives of management. The Leopold Report was generated by a wildlife controversy in Yellowstone National Park, but its chair, A. Starker Leopold, broadened the report to a grand vision of the purposes of national parks (Leopold and others 1963).

The report recognized that the primitive landscapes of America were, in large part, products of disturbance, including fire, and that in the long run, management would only be successful if it was to manage these disturbances, rather than just suppress them. The authors were somewhat pessimistic that this could ever occur, but dreamed of recreating the "...vignette of primitive America...at least on a local scale." The report was very radical for its time and was circulated by the DOI for a year before Secretary of the Interior Udall accepted it. That year, 1964, was the same year the Wilderness Act passed and was signed into law by President Johnson. It defined wilderness as an area "...untrammeled [unaffected] by man," "...affected primarily by the forces of nature..." and "...managed to preserve its natural conditions." The Leopold Report and the Wilderness Act provided similar guidance to scientists and managers. Clearly, the natural force missing from almost every park and wilderness area was fire: How could it be reintroduced to these systems? There was no regulatory guidance for an operational application of fire management until 1968, when the DOI released its new fire policy, based on the concepts of the Leopold Report. This new policy not only recognized prescribed fire as a legitimate action, but also sanctioned the use of natural fires where appropriate.

Within the same year, a fire management program was instituted at Sequoia and Kings Canyon National Parks, accompanied by a research program that investigated the effect of these programs on fuels, flora and fauna (Kilgore and Briggs 1972). Yosemite National Park followed in 1970. These parks had primarily low- and moderate-severity fire regimes (c.f. Agee 1993), where fire historically was fairly frequent and few of the fires were of stand-replacement intensities over large areas (mixed-conifer/pine, red fir). Higher-severity chaparral areas were avoided in the initial years. The broad granite terrain of these parks also helped contain fires to individual valleys: Long wind-driven intense fire runs were uncommon there. The early research there (Agee 1973; Biswell 1961, 1967; Hartesveldt 1964; Kilgore 1971a,b, 1972, 1973; Parsons 1976, 1978; van Wagtenonk 1972, 1974, 1978) clearly showed that prescribed fire could be valuable in moving ecosystems back to more natural conditions, without unacceptable resource damage, and that

prescribed natural fire could be successfully managed (Kilgore and Briggs 1972). Although Forest Service research had been helpful to the NPS scientists and managers, in both research and application the NPS was a leader by the early 1970s (van Wagtenonk 1991a).

Yellowstone National Park began a prescribed natural fire program in 1972 (Romme and Despain 1989). Research and monitoring there found two seemingly apparent patterns: (1) Fires tended to burn primarily in old-growth forest (Sweaney 1985) and naturally extinguished themselves at the boundary of younger forest; and (2) very large fires were characteristic of the distant past (Romme 1982). Romme's work was somewhat consistent with the monitoring, in that he found older forest to be more flammable than younger forest. But his reconstruction of the Yellowstone landscape since the early 1700s suggested an ecosystem never in equilibrium or stability at any park scale, due to large events at infrequent intervals. The implications of these findings were never addressed by the fire management plan for Yellowstone, although they were available almost a decade before the fires of 1988.

The Forest Service began a similar wilderness fire program in the Selway-Bitterroot Wilderness in northern Idaho in 1972. This area contained forest types in moderate- and high-severity fire regimes (Brown and others 1995b), and the second fire that was allowed to burn (Fritz Creek 1973) escaped, burning about 500 ha outside of the management unit (Daniels 1974). The fire had been monitored during the burn, and research work was initiated after the smoke had cleared (Mutch 1974). The program was continued, although it was later described by the agency as meeting with "moderate" success (Towle 1985). In 1978 the Forest Service adopted a nationwide "appropriate response" suppression strategy that more clearly allowed this type of integrated fire management. A naturally occurring ignition, under this policy, could be declared a wildfire, but limited resources might be directed to suppress it. Manager-ignited prescribed fire was not allowed in designated Forest Service wilderness through the mid-1980s.

The adoption of wilderness fire management plans that incorporated prescribed fire or prescribed natural fire blossomed in the 1980s. Associated with this increase were extensions of plans into primarily high-severity fire regimes and the increase in both prescribed fire and prescribed natural fire (Botti and Nichols 1995). Management was clearly moving faster than research, partly because of limited funding for park and wilderness research, and the limitations of science to address operational concerns.

Limitations of the Science Through the Mid-1980's

The primary limitation posed by science for wilderness until the fires of 1988 was the dissolving paradigm of successional theory. The fading of a firm theoretical model (classical Clementsian climax theory) to apply to disturbance in natural ecosystems allowed managers to view reintroduction of fire as a "good" thing without much attention to either what fire was doing or where it might go. Ecological problems with some fire programs were difficult to solve because of a lack of records on where burns occurred

and a lack of monitoring of the fires' effects on resources (Thomas and Agee 1986).

The classical view of shifting paradigms (Kuhn 1970) was that after an accepted model of science (a paradigm) was created, evolving research would accumulate evidence suggesting the current paradigm was too simple or just wrong. Eventually, a relatively rapid shift towards a more robust model would occur, and that new paradigm, in turn, would eventually be rejected in favor another, more robust paradigm. In plant ecology, the major paradigms of the century themselves underwent a succession similar to initial floristics (Egler 1954; Agee 1993), where many of the species (theories) represented in the successional sequence are present in early succession but display differential dominance over time. The major plant ecology theories were all proposed within a decade early in the 20th century, but exhibited differential dominance over time.

The classical view of plant succession (the theory that attained initial dominance) persisted much of the 20th century: the Clementsian view of regional convergence towards a vegetation life-form created by autogenic succession in the presence of stable climate (Christensen 1988, 1991). Although competing models were proposed early (Gleason 1917, Tansley 1924), the Clementsian model was not seriously challenged until Odum (1969) proposed an ecosystem model that had a number of tautological premises. Among them were assumptions that diversity and stability increased with ecosystem development (time since disturbance). Odum's paper generated a number of rebuttals (such as Drury and Nisbet 1973) that suggested that ecosystems did not have emergent properties, that various forms of diversity might peak in early succession and that stability might in some cases be maintained by disturbance. Rather than producing a more robust paradigm, these challenges to the existing order recognized that ecology is a science of place and time. Grand unified theories are unlikely to apply (Christensen 1988). Much of the new theory was developed by ecologists who had worked in disturbance-prone ecosystems, and they recognized the multiple pathways that succession might take after disturbance, a function of both the disturbance and the "players" or organisms at the site. Disturbance, rather than a binary presence-absence variable, became a complex combination of characteristics (White and Pickett 1985).

Wilderness fire scientists welcomed these challenges to the classical theory. The incorporation of disturbance into new theory provided a scientific niche for the presence of fire in wilderness: Disturbance had a place in natural landscapes (White 1979). It was now possible to more clearly explain the previously baffling myriad of successional trajectories after disturbance. But as the challenges were comforting in one sense, they were discomfoting in another. To what the new theory added in recognizing fire as a natural factor, it removed in discarding the notion of convergence toward stable ecosystem states (Christensen 1991). This created two managerial challenges: (1) The issue of what to preserve became much more complex, as ecosystem classification resulted in much less convergence of community types; and (2) The stable end point toward which we should manage suddenly disappeared, leaving managers groping for a definition of a natural ecosystem state or states. This latter point had crucial significance for wilderness fire.

This question took form in 1980s wilderness as a debate between structure and process as appropriate goals for park and wilderness management. In a somewhat simple synopsis, the process argument stated that every past landscape was a snapshot of a variable ecosystem, and that ecosystem would vary into the future. Reintroducing the process of fire would eventually restore an uncertain but natural future set of ecosystem states (Parsons and others 1986). This view was supported by some of the early interpreters of the Wilderness Act (Worff 1985a,b). The structure argument (Bonnicksen and Stone 1982) stated that in any ecosystems where an unnatural structure had developed, reintroducing fire without attention to current structure could not result in a restored natural ecosystem. To some extent, the debate depended on where one was (Agee and Huff 1986): after all, ecology is a science of place. But the question remained even where scientists were viewing the same place. The argument became most heated in the Sierra Nevada/Cascades low-severity fire regimes, where almost everyone agreed on the degree of ecological change but differed on the need for structural approaches to restoration (Bancroft and others 1985; Bonnicksen 1985).

Added to the uncertainty of a desired future condition was the uncertainty of the disturbance regime. In the 1960s, the recognition of fire as a natural factor was sufficient to encourage management implementation. In the 1970s and 1980s, more information began to emerge about fire regimes. White and Pickett (1985) defined a number of characteristics important for understanding the effects of disturbance (such as frequency, magnitude, seasonality, extent, etc.), but for fire regimes, the primary one investigated was frequency, and primarily for low-severity fire regimes. Kilgore's review of wilderness fire (1986) for the first conference on wilderness focused primarily on frequency within broad fire regime types. More than 40 references to fire frequency were made by generalized fire regime types. The fire regime types did carry implications for fire intensity, but little was known about extent, season or synergism with other disturbances. Variability and patchiness, now known to be very important, were largely unquantified. Some information on variability in fire frequency was presented in terms of ranges of fire frequency. Complex fire regimes in the moderate severity fire regimes had little information available on patch size, proportions of different severity or other aspects of the fire regime.

Standards for monitoring were largely lacking during this period. Success was often gauged by area burned by prescribed fire and/or prescribed natural fire. Even though uncertainty about the operational goals of fire management (fuel reduction, ecological effects, etc.) persisted, there was little information that could be used to track progress towards any goal. Concerns about visual effects of prescribed fire in giant sequoia groves led to establishment of an independent committee to review the fire program at Sequoia and Kings Canyon National Parks (Christensen and others 1987; Cotton and McBride 1987). The committee recommended development of a detailed monitoring system for fires by the National Park Service.

Stand-level dynamic models incorporating disturbance began to emerge in the 1970s, but they suffered from the absence of established subroutines for stand growth, fire effects, or fire behavior. Most were derived from the JABOWA-type

gap models that grew stands on a small area (Botkin and others 1972). The first model, FYRCYCL, was developed at Yosemite (van Wagtenonk 1972) and was far ahead of its time in using historical fire weather to drive the fire portion of the model. Another early model was SILVA (Kercher and Axelrod 1984), which was an improvement on FYRCYCL in the stand growth routine but less elegant in its fire behavior and fire weather. Fire effects on trees were estimated from scorch height (a function of fireline intensity) and tree diameter. However, many of the weather inputs were held constant, so a crude simulation at best of the fire regime was possible. CLIMACS (Dale and Hemstrom 1984) was another fire model parameterized for the Pacific Northwest. Its stand growth subroutines were robust but it treated disturbance as an external effect that required the user to define exactly which size classes and species were removed from a particular disturbance. It was verified for only one forest type in the region.

Two models linking fire behavior and fire effects were developed during this period. Peterson and Ryan (1986) developed an algorithm that integrated stand-level characters and fire behavior (including estimated flame residence time) into a probability of mortality that was a function of volume of crown kill and the ability of a given bark thickness to withstand lethal heat. The model requires estimation of burning time in order to compare time of lethal heat to critical time for cambial kill (based on bark thickness), and burning time was not commonly available to users. Ryan and Reinhardt (1988) used empirical data to develop a similar mortality function based on crown scorch volume and bark thickness.

One of the major developments useful in fire behavior analysis was adaptation of the Rothermel spread model (1972) to a variety of stylized fuel models (Albini 1976), including those applicable to wilderness. A PC-version known as BEHAVE was made available in 1984 (Burgan and Rothermel 1984), with later improvements in several areas (Andrews 1986). This model allows prediction of surface fire behavior for given fuel, weather and topographic predictions. At high levels of input variables, fire behavior expressed as fireline intensity or flame length can be interpreted as leading to erratic fire behavior, but crown fire models during this period were limited to empirical studies in boreal forests (Van Wagner 1977).

Most of the growth in operational fire management plans in the 1980s was in parks and wilderness areas with moderate- to high-severity fire regimes, suggesting that these plans contained sufficient research information on effects and behavior of fire to indeed make these "prescribed" natural fire plans. In most cases, this information was very generalized. Boundaries of prescribed natural fire zones were rather arbitrarily drawn inside the boundaries of the preserves, with little attention to the main direction of spread for intense fires or their historical or projected eventual size. Historical size could be estimated from fire history research, but technology to project fire behavior days or weeks in advance was not available. In other areas, such as the chaparral of California, research in high-severity fire regimes did occur but focused on ecological effects of fire (Baker and others 1982; Parsons 1976; Rundel and Parsons 1979, 1980) and much less on behavioral aspects. Limited research in the Pinnacles Wilderness

(Agee and others 1980) focused more on behavior than ecology.

Social science research was encouraged during this period, focusing on visitor perceptions and acceptance of wilderness fire. Visitors who understood the role of fire in wilderness generally supported the policies (Cortner and others 1984; Rauw 1980; Stankey 1976; Taylor and Daniel 1984; Taylor and Mutch 1986). The economics of fire in wilderness remained clouded due to the blending of fire management activities outside and inside wilderness which made separation of costs difficult, and the different ways that agencies accounted for prescribed natural fire versus wildfire in the pre-Yellowstone fires era. The Forest Service and some regions of the National Park Service required upfront budgeting for monitoring activities; when that budget was expended, the fire was reclassified as a wildfire (Agee 1985, Daniels 1991). Another complication is the contrast between classical "least-cost-plus-loss" approaches, which assumes all resource change is a loss, and evaluation of resource change when fire could be viewed either as a cost or benefit. Mills (1985) defined the major obstacle to appropriate economic analysis of fire in wilderness as understanding the "natural state" objective of wilderness which would then allow resource change to be viewed as cost or benefit. Ecologists, as noted above, had been little help in agreeing on a consensus definition useful for economic analysis.

The Wilderness Fire workshop held in Missoula in 1983 (Brown and others 1985) defined the major issues apparent at that time. Over 100 papers and posters were presented at the conference, and five major issues were addressed: (1) the "natural fire" issue—what is natural; (2) the "Indian fire" issue; (3) the "lightning (prescribed natural fire) versus human (prescribed fire)" issue; (4) the "fire size and intensity" issue; and (5) the "unnatural fuel buildup" issue. There were no resolutions of these issues at that time, but considerable discussion of each. Clearly, the issue of "naturalness" was paramount in the first three topics. Are the origins or effects of fire the basis for "natural?" Native Americans burned many of the landscapes of their day, often repeatedly, and these effects had a large influence on vegetation as far back as we can reconstruct it (Arno 1985; Gruell 1985; Kilgore 1985; Lewis 1985). How should this be incorporated into current fire planning for wilderness? The lightning versus human ignition issue is tied to the previous questions and to the last question as well. Arguments about how close a prescribed fire can mimic a natural ignition (Despain 1985), the need for caution in using prescribed fire in wilderness (Daniels and Mason 1985), the need to focus on fire effects (Van Wagner 1985) and the need to keep human hands off wilderness (Worff 1985a) all surfaced in this discussion. The management-caused fuel buildup in some ecosystems was suggested to be reason enough for prescribed fire programs to restore more natural conditions (Brown 1985; van Wagtenonk 1985).

Yellowstone: The Revolution of 1988

A revolution is defined as a drastic change of any kind, and that describes the events of the summer of 1988. Yellowstone's fires were at the center of the controversy because of their

visibility, but other fire events occurred that same year under similar circumstances.

Yellowstone's Fire Program

Yellowstone's prescribed natural fire program began in 1972 and was considered by the Park to be a successful program before 1988. An average of 30 fires per year burned between 1972 and 1987 (Despain and Romme 1991), and about half were monitored. The monitoring of the fires during this time indicated that fuels were a major determinant of where fires burned, with weather influencing the behavior of the fires. Most fire starts and fire spread occurred in older lodgepole pine (*Pinus contorta*) stands, and fires appeared to naturally extinguish themselves at the edges of younger stands. The monitoring results might have been interpreted to mean that as more natural fires burned, the Park would be buffered from extreme events by the patch mosaic of fuels (Sweeney 1985). However, work by Romme (1982) had suggested that a very large event had occurred in the early 1700s over at least part of the Park.

The summer of 1988 brought many fires and little precipitation compared to the 1972-1987 record, a very short period of comparison for a high-severity fire regime of hundreds of years. It is not surprising that conditions of the extreme event were not forecast, and two-thirds of the 1972-1987 period July and August precipitation was well above long-term averages (Despain and Romme 1991). When the fires of 1988 began to spread, they were pushed by a series of cold fronts, which resulted in substantial increases in fire area in short periods of time, capped by the runs of early September that resulted in fire area growth of tens of thousands of ha per day.

By the end of the summer, over 300,000 ha (750,000 ac) of the Park, and similar areas around it, had burned in a spectacular series of fire runs. Roughly half of the area burned was from direct or indirect human causes (camper, firewood, power line), reviving the argument of whether nature cared who started the fire (Van Wagner 1985). Park researchers defended that area as "natural" by claiming that natural fire starts in each area occurred later in the same year and, under the extreme conditions of 1988, would have resulted in similar spread patterns (Despain and Romme 1991). Yet that argument remains a weak *ex post facto* attempt to justify the argument that we were witnessing a "natural" event of unparalleled magnitude in recent history. Certainly the scale had precedent (Pyne 1982), but human activities altered the pattern and extent of the fires of 1988 (Christensen and others 1989).

Canyon Creek

The Canyon Creek fire burned in the Bob Marshall Wilderness. Ignited by lightning on June 25, 1988, it was designated a prescribed natural fire and was allowed to burn (Daniels 1991). It stayed at less than 1 ha (2.5 ac) for 26 days, but in late July grew to 4,000 ha (10,000 ac) in three days, burning in a mosaic pattern so that about a third of the encompassed area actually burned. After 65 days of active management, the fire escaped the wilderness boundary and grew from about 25,000 ha (60,000+ ac) to almost 100,000 ha (250,000 ac) in 16 hours, at the same time the Yellowstone

fires were rapidly expanding. Full suppression action was ordered for the fire.

Prophecy Fire

The Prophecy fire burned at Crater Lake National Park, Oregon, in August 1988. It began in the eastern boundary area of the Park, but was within the approved natural fire zone. Crater Lake had managed natural fires for a decade in the moderate-severity red fir type, and these burns had remained in prescription. The Prophecy fire was pushed by strong westerly winds and moved out of the Park to cover about 400 ha of Forest Service land to the east. These winds may not have been unusual, but the absence of weather stations in the area meant that this fire weather, and the associated fire behavior, would not be predicted. The fire crowned through a sparsely vegetated climax lodgepole pine type that was thought to rarely support such behavior (Agee 1981, Gara and others 1985).

Sifting Through the Ashes

By late summer of 1988, the political climate of an election year, combined with the perceived multi-regional, multi-agency failure of the natural fire program, resulted in the suspension of all such programs until completion of a review and implementation of any review recommendations. Local policy reviews of the Yellowstone situation (Christensen and others 1989) and a major national fire policy review (Philpot and Leonard 1989) were completed before the end of the year. The local review focused on ecological issues and proposed both research and management recommendations for Yellowstone. For research, the review recommended an ecosystems approach, a landscape or geographic context for individual projects and provision for long-term studies (Christensen and others 1989). For management, the local review recommended that an ecological blueprint evolve on a wilderness-specific basis, to articulate clearly the range of landscape configurations locally acceptable and to guide fire management planning. The national review (Philpot and Leonard 1989) suggested that the natural fire policy was in general a sound policy, but that it had been implemented without sufficient prescription criteria. Most of the plans that did not meet current policy were in national parks (Wakimoto 1989).

The Flame Flickers: Politics and Philosophy After Yellowstone

The political landscape has been as important as the natural landscape in directing wilderness fire science. The events of 1988 essentially shut out wilderness fire, and the recovery of management programs over the past decade has been relatively slow. No one wanted to be the supervisor of the next Yellowstone event. Some wildernesses, such as Yosemite and Sequoia-Kings Canyon, which pioneered both prescribed fire and prescribed natural fire, had their programs reinstated almost immediately, as they met the criteria of the 1988 national fire policy review even before 1988. Other suspended programs have never been reinstated. The result was a significant and immediate decline

in numbers of fires and area burned (fig. 1; Parsons and Landres 1998). Although area contained with prescribed natural fire zones increased by seven percent between 1988-92, area burned by prescribed natural fires decreased by 94 percent (Botti and Nichols 1995), largely due to conservative management criteria, including funding. At the same time, prescribed fire activity doubled over its pre-1988 levels (Botti and Nichols 1995), but this is largely due to increases for one unit (Big Cypress National Preserve).

The conservative management criteria were all based on control (flame length) or external issues (smoke, availability of regional forces). Not a single criterion was based on meeting objectives for wilderness management. Given that planning context, major reductions in numbers of programs and fires allowed to burn are not at all surprising. Yet the operational management plans were not to blame. Without an ecological blueprint for what was desired in wilderness, it was not only much easier but more defensible to define conditions where fire was not wanted than to define conditions where it was.

The consolidation of research scientists in the Department of the Interior also affected wilderness fire science. The management agencies (such as the NPS) lost their ability to fund research, because that function was now in the newly created National Biological Survey. The brief life of both the National Biological Survey and its replacement, the National Biological Service, resulted in financial chaos for research scientists, and funding for fire research has continued to be problematic in the Geological Survey, where these scientists now reside.

The political developments and problems of wilderness fire management began to erode the "era" of wilderness fire (Pyne and others 1996). Pyne correctly foresaw the 1990s as a new era of urban intermix fires, and it was ushered in with the horrific Oakland fire of 1991 (Ewell 1995). Pyne's declaration was rooted in the belief that the philosophical questions posed by the marriage of fire and wilderness had never been resolved and that technical approaches could not resolve them. Yet in the end, technical approaches must be employed to foster operational fire management programs, even if the philosophical issues remain unresolved.

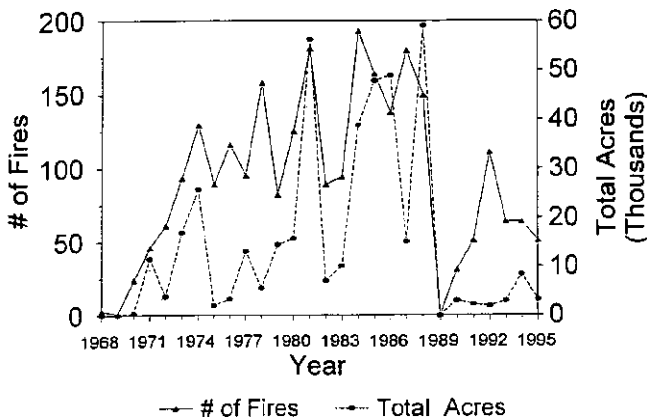


Figure 1—Trends of numbers of fires and area burned since the inception of prescribed natural fire programs in 1968. Note the pronounced drop after 1988 (Parsons and Landres 1998).

Science Since Yellowstone

The science of wilderness fire has progressed remarkably in the past decade, withstanding the political issues and a largely fragmented research approach. There have been few large research programs directed specifically toward wilderness fire, partly because of the fragmented, multi-agency management of wilderness and a lack of research focus that is characteristic of many other large, national-scope projects (Long Term Ecological Research, International Biological Program, NASA's space program, etc.). The NPS Global Change program is one larger program that has produced some substantial implications for wilderness fire. Yet many of the technical developments have resulted from locally funded projects, or from research done for other purposes.

Drivers of Wilderness Fire

That fuel, weather and topography drive the behavior of an individual fire has long been known (Barrows 1951, Brown and Davis 1973). Yet the factors driving wilderness fire regimes continue to be debated: Are fuels or weather more important? Our research of the past decade suggests that the answer not only differs by fire regime, but to some extent on the interaction of fuels and weather. Swetnam and Betancourt (1990) linked a set of regional cross-dated fire histories in ponderosa pine (*Pinus ponderosa*) forests to high (La Nina) and low (El Nino) phases of the Southern Oscillation. During the El Nino phases, precipitation in the Southwest is much higher and fire activity is much less. At the same time, tropical and subtropical areas receive less precipitation as those storms are moving further north. Large areas burned in Borneo (Davis 1984) and Australia (Rawson and others 1983) during a large El Nino event in the early 1980s. This link between global climate and local variability in fire regime shows a trend that links wilderness to the rest of the world.

In high-severity fire regimes, arguments about the relative influence of fuels and weather continue (Weir and others 1995, Wierzchowski and others 1995). In Canadian boreal and subalpine forests, prescribed fire has been used operationally under the assumption that decades of fire exclusion have changed these forest types, that younger stands have not been created during that period and that older forests were more flammable. Bessie and Johnson (1995) concluded that weather was the primary driving factor in large fire behavior; and since large fires constitute almost all the area burned, fuel conditions are relatively unimportant. They generalized these conclusions to all forest types, a conclusion rebutted by Agee (1997). He suggested that under extreme weather in low-severity fire regimes, fire size may well have increased, but that fire severity may not have been markedly increased. Fuel conditions have been shown to affect fire behavior and extent in low- (Wright 1996) and moderate-severity (van Wagtenonk 1995) fire regimes (fig. 2).

In some high-severity fire regimes, fire return intervals may be so long that very unusual synergistic influences may occur and mask more simple correlations of fire with flammability-stand age or weather-climate patterns. In the Olympic Mountains, Henderson and others (1989) mapped a very large forest fire event (fig. 3) circa 1700 A.D. that had been

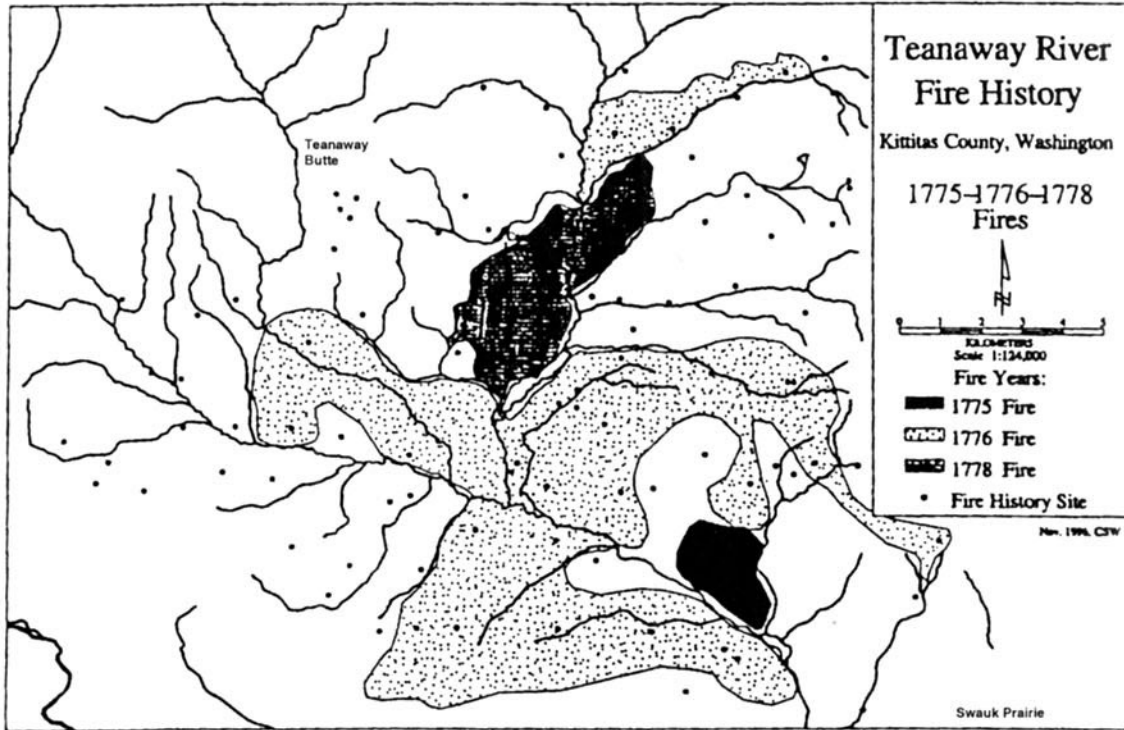
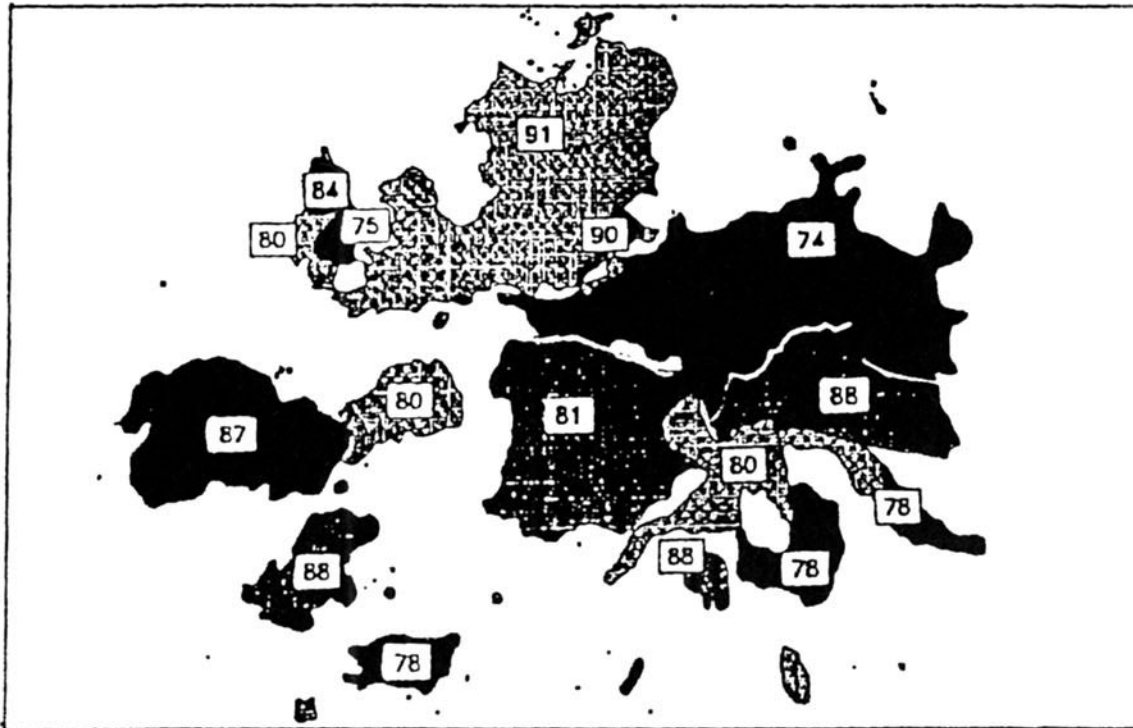
A**B**

Figure 2—A. Reconstructed fires of 1775-1778 in mixed-conifer forests of eastern Washington (Wright 1996). Fires occurring with 1-2 years of one another in this low-severity fire regime appear to be extinguished when they enter recently burned areas. B. Monitored fires 1974-1991 in Yosemite National Park show similar mosaics (van Wagtenonk 1995). These appear to be more stable patterns than in high-severity fire regimes where process overwhelms pattern under severe weather (Romme and Turner 1991).

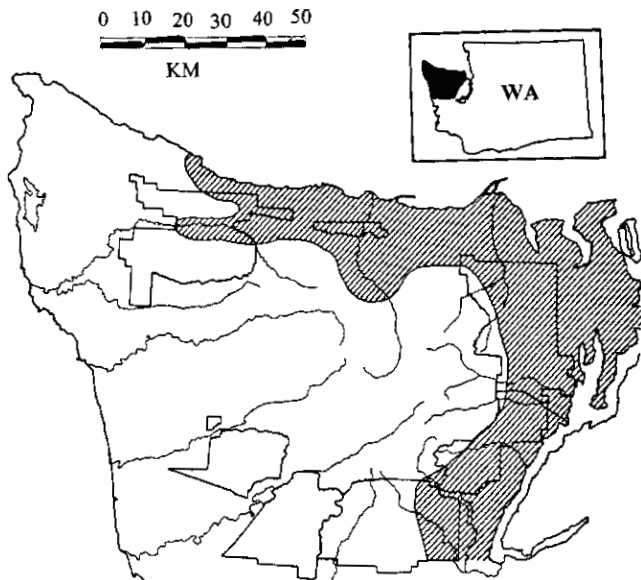


Figure 3—The large fire of ca. 1700 in the Olympic Mountains (Henderson and others 1989) appears to have occurred after a large earthquake in 1700. This quake, occurring in January, may have been associated with considerable treefall and copious dead fuel, needing only an ignition source to become very large.

partially identified by Fonda and Bliss (1969). Fire cycle models based on climate (Agee and Flewelling 1983) were unable to reproduce similar fire events, and it was thought that very unusual patterns of lightning frequency or foehn winds may have occurred in the past. Recently, a very large historic earthquake along the Washington coast was reconstructed from tree ring records of trees buried beneath sea level by submergence of coastal lands at the time of the quake (Yamaguchi and others 1997). This date was consistent with records of a major tsunami that hit the east coast of Japan on January 26, 1700. At a time when soils are saturated, this earthquake likely felled many stands of trees around the peninsula, and this additional dead fuel may have driven the large fire activity that apparently occurred. Lightning frequency, drought or foehn winds, the usual combinations of factors associated with large fires, may have remained quite average during this period.

Where fire return intervals are quite long, these “surprises” may be a major factor in the disturbance dynamics. Not only may extreme events be driving the system, but they may be events that we have not yet uncovered. Lertzman and others (1998) showed through simulating fire regime parameters that substantial variability may result, even in the absence of an underlying physical or ecological pattern. They recommend caution in attributing causality of fire regime drivers that are not motivated by independently generated hypotheses.

Fine-Tuning the Fire Regime

When early fire management programs began in wilderness, general knowledge of the fire regime was considered adequate. Research inside and out of wilderness has led to a

more precise understanding of the fire regime, but it is still not possible to generate many parameters of a fire regime by simply knowing, for example, what forest type is being considered. Where more precise information has been generated, it usually shows variability in frequency, intensity or extent. Synergistic effects are known to be more important than previously considered, although our ability to predict them is still poor. And the general implications for management have been clouded by the complexity of these emerging fire regimes. Faced with considerable ranges in variability, which combination is appropriate for a certain place now? Research on fire regimes has allowed us to place bounds on uncertainty, but it has also generally driven us away from relying on simple statistics like the mean. Programs have evolved from rather uniform burns to those incorporating considerable variability (Bancroft and others 1985; Parsons and Nichols 1986).

Fire frequency has always been a primary parameter of the fire regime. Kilgore’s wilderness fire review (1986) has over 40 citations on fire frequency in selected wilderness ecosystems, and he recognized that more examples could be cited. But information on other fire regime parameters was lacking. Since that time, we know even more about fire frequency in wilderness. These new data have allowed us to understand the distribution of fire frequency, not just its central tendency. A remarkable achievement was the reconstruction of giant sequoia (*Sequoiadendron giganteum*) fire regimes back over millennia (Swetnam 1993). The mean fire-return interval shifted significantly for this low-severity forest type over periods of centuries, and inferences about fire intensity were made from correlations of tree-ring growth with fire occurrences and percentages of sample trees scarred from an individual fire. Landscape juxtaposition of forest types was found to be important in determining fire frequency. In the north Cascades, where wet, west Cascades forest types are mixed with dry, east Cascades types due to a rainshadow effect west of the Cascade crest, the wet types had fire-return intervals well below those measured elsewhere in the Cascades for those types. The dry, eastside forest types had fire return intervals well above those measured in the eastern Cascades (Agee and others 1990).

Fire intensity remains difficult to reconstruct from historic fire regimes. Reconstruction of growth on trees experiencing fire, and defining age classes of trees likely to establish in fire-generated gaps, have been used to infer historic intensities. In giant sequoia groves where the history of prescribed fire includes some fairly hot burns, reconstruction of tree-ring growth showed that fire generally increased growth, but some variable response was evident (Mutch and Swetnam 1995). A delayed growth response was found where very intense fires had occurred and scorched the foliage of the sequoias. Sequoia regeneration was tied to fire-generated gaps where sunlight could penetrate to the forest floor. These data were used to infer past fire intensities. For example, a fire in 1297 A.D. was inferred to be relatively intense due to the increase in tree growth on giant sequoias (fig. 4), suggesting a release from competition and substantial regeneration that occurred locally (Stephenson and others 1991). A recent article suggests that high-intensity fire also was characteristic of ponderosa pine stands (Shinneman and Baker 1997). However, these stands in the Black Hills are transitional to boreal forest; white spruce

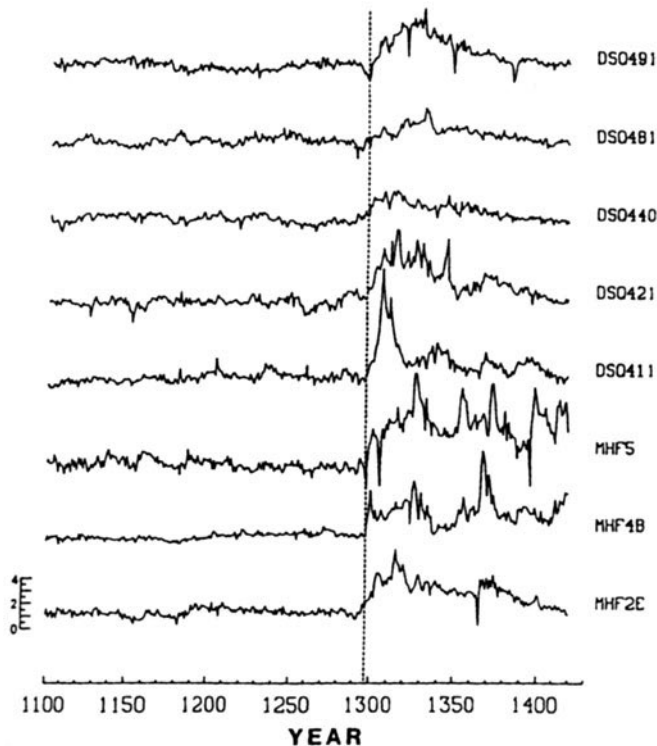


Figure 4—Patterns of tree-ring growth from giant sequoias (Stephenson and others 1991) show a pronounced growth effect after a reconstructed fire of 1297 A.D. Unlike many previous fires, this one must have been severe and reduced competition, as all trees show a growth release. The pattern of unusual growth continues for a century, and some fires are associated with decreases in growth for sample trees. This suggests severe fires did occur in sequoia groves, and reminds us of the variability in fire regime for very long-lived organisms like giant sequoias.

(*Picea glauca*) is a common understory species, and a complex mix of fire regimes (e.g., Agee and others 1990) should be expected where types are in transition. Tree regeneration is closely linked to fire severity; in moderate-severity fire regimes, severity will have significant effects on tree species likely to establish (Chappell and Agee 1996).

Quantifying season of burning has been important because of the opportunity to ignite prescribed fires over a broad seasonal range. What is most natural? Historical seasonality has been evaluated primarily for low-severity fire regimes by defining the placement of the fire scar for a particular year in the earlywood to latewood of the annual ring. In Southwest ponderosa pine stands, most scars are in the earlywood, defining spring as the most common season for fires (Baisan and Swetnam 1990), although some areas exhibit more even distribution of fires across the growth season (Grissino-Mayer and Swetnam 1995). In the Pacific Northwest, the same species exhibits mostly late-season fires (Wright 1996). Heyerdahl (1997) showed that there was considerable seasonal variation in the Blue Mountains of Oregon and Washington. Southerly Blue Mountain stands had a longer snow-free season and more scars within the growing portion of the annual ring than stands of the same species composition in the northern Blue Mountains, which had a shorter growing season and a concentration of scars after growth for the year had ceased.

Synergism, or the interaction of fire with other disturbances, was recognized by White and Pickett (1985) as an important parameter of disturbance regimes. Very little quantification of this effect was evident for fire regimes before the late 1980s. Interaction with insects has long been recognized as a major second-order fire effect (Fischer 1980), but defining the degree of interaction is difficult, as many other factors are important (Amman and Ryan 1991). After the Yellowstone fires of 1988, the major tree species in the area (lodgepole pine, Douglas-fir [*Pseudotsuga menziesii*], Engelmann spruce [*Picea engelmannii*], and subalpine fir [*Abies lasiocarpa*]) were attacked by a variety of insects; between 28-65% of the trees living after the fire were infested and killed (Amman 1991). Most of the bark beetle-attacked trees had basal damage from the 1988 fires.

At Crater Lake National Park, Swezy and Agee (1991) found that low-intensity but long-duration fires, caused by forest floor buildup due to fire exclusion, killed many of the fine roots after late spring burns. Low vigor, old-growth pine trees had an increased level of insect attack and mortality after these fires, and fall burning was recommended as a better season, based on surveys of trees burned in spring and fall.

Disease can also be an important synergistic factor. In the western United States, perhaps the most important synergism between fire and disease is the introduced white pine blister rust (Kendall and Arno 1990). This disease causes cankers on the stems of young pines and kills them. When fire kills older trees, recolonization of whitebark pine (*Pinus albicaulis*), often mediated by Clark's nutcrackers (*Nucifraga columbiana*) (Tomback 1982) may be short circuited. In mountainous terrain, snow avalanches can create persistent snow avalanche paths and alter other processes such as landsliding and future fire spread (Butler and others 1991).

Models

The past decade has witnessed an explosion in personal computing power and with that growth, an accompanying expansion of models attempting to explain fire behavior and effects. These models have particular relevance to wilderness fire because they allow forecast of spatially explicit fire sizes, as well as fire effects.

One of the more important models for fire effects has been the individual tree model FOFEM (First Order Fire Effects Model; Reinhardt and others 1997). It scales mortality to the stand level by aggregating individual tree effects to the stand level based on the Ryan and Reinhardt (1988) mortality algorithm. This model has gone through four iterations in the past decade and will continue to be updated periodically. It is national in scope and provides information in addition to tree mortality on fuel consumption, mineral soil exposure and smoke. Synergistic effects, which tend to be difficult to predict as second-order interactions, are not predicted by FOFEM. Nevertheless, it has served as the basis for tree mortality prediction in several important models.

A variety of individual-based gap models have been developed since the 1970s (Hinckley and others 1996; Urban and others 1991), but few have concentrated on incorporating fire. FIRESUM (Keane and others 1989) was an improved gap model that incorporated stand growth and disturbance

for inland Northwest conifers. The fire algorithms were complex, but the stand-level results were greatly influenced by the initializing stand condition; an individual tree dying of old age, for example, had a large influence on the basal area output over the simulation period.

While the science of gap modeling grew, the ability to represent wilderness landscapes in geographically referenced form also increased. Geographic information systems (GIS) represented a way to evaluate often inaccessible landscapes in digital form. The development of better software packages and more powerful personal computers allowed robust analyses to occur at relatively low expense. Fire applications, such as analysis of historic fire incidence by vegetation type, fuel inventories, prescribed burn units, lightning strike incidence analysis and fire regime analysis, were done (van Wagtenonk 1991b). Links of these types of analyses to fire growth simulators were beginning (Bevins and Andrews 1989). Development of accurate input layers for the current generation of fire area growth models remains relatively poor (Keane and others 1998).

FIRE-BGC (Keane and others 1996a) was developed by marrying some of the algorithms of FIRESUM with FOREST-BGC, a physiologically based model (Running and Gower 1991) that has been scaled up to a landscape approach. As applied to wilderness ecosystems in Glacier National Park (Keane and others 1996a) and the Bob Marshall Wilderness (Keane and others 1996b), the model links many across-scale interactions, but it has the universal problem of marrying not only diverse spatial scales, but those of time as well (Keane and others 1996a). Temporal information at scales from annual (stand growth equations) to hourly (fire growth equations) complicate current modeling efforts.

Disturbance propagation across landscapes has been modeled in two general ways: percolation-type models and deterministic models. The percolation models suffer from the fact that fire does not move across a landscape with equal probabilities of spread in all directions. The deterministic models suffer from data deficiencies (Van Wagner 1987). Both have increased our knowledge of fire effects and behavior at broader scales.

The percolation models have increased our knowledge about the influence of landscape pattern on process (fire) (Turner 1989). Most of the percolation work has been in high-severity fire regimes, where the binary process of a cell being occupied or not by disturbance fits the high-severity nature of the disturbance. Work in the 1980s suggested that disturbance in heterogeneous landscapes was dependent on the structure of the landscape, as well as disturbance frequency and intensity (Turner and others 1989). This evolved to a more complex view that disturbance probability affecting percolation can change over time, particularly where fire weather becomes extreme (Turner and Romme 1994). Under extreme conditions, process is relatively independent of pattern (Agee 1998; Romme and Despain 1989). Nonequilibrium systems will be the result (Baker 1989, Turner and Romme 1994); scientific advances in landscape theory have resulted in a tougher job for managers by increasing the envelope of uncertainty. Percolation-type models have suggested that landscapes altered by past intervention in fire regimes, or those subject to climate change in the past (for example,

Clark 1988) or the future, will take 0.5 to 2 rotations of the new disturbance regime for the landscape to adjust to that new regime (Baker 1989, 1994).

In contrast to the ecological gap and disturbance models, fire behavior models received less attention over the same period, yet our inability to predict fire spread and intensity has had much more effect on wilderness fire programs than imprecision in predicting ecological effects. Fortunately, substantial progress has been made in landscape modeling of fire behavior. A nonspatial model (RERAP) was developed to determine probabilities that a prescribed natural fire would exceed an acceptable size (predetermined by the user) before a fire ending event (precipitation) would halt spread (Carlton and Wittala, no date). However, it has not been widely used in wilderness fire management. A fire growth simulator (Bevins and Andrews 1989) was developed by the Forest Service, and a similar model was being developed by the National Park Service (Finney 1995). These efforts merged in the mid-1990s at the Missoula Fire Sciences Laboratory.

The model currently holding most promise for wilderness fire behavior is FARSITE, a spatially and temporally explicit fire growth model (Finney 1998). The model was initially developed to help predict spread of wilderness fires, but it has shown great applicability to wildlands in general. The landscape “themes” or data layers require information on elevation, aspect, slope, fuel model and canopy cover, with optional themes for crown fire behavior: crown height, crown base height and crown bulk density. Daily and hourly weather streams are required over the simulation period. Surface fire, spotting and crown fire behavior are simulated, subject to the limitations of models that currently exist for those types of fire behavior. Fires spread in the model using Huygens’ principle, where the fire front is expanding based on elliptical wavelets, the shape of which depends on the fuel model and local wind-slope vectors (fig. 5). Backing and flanking fire spread is estimated from the forward rate of spread, as the current fire spread model (Rothermel 1972) only predicts the forward rate of spread. Finney (1998) discusses the limitations of FARSITE.

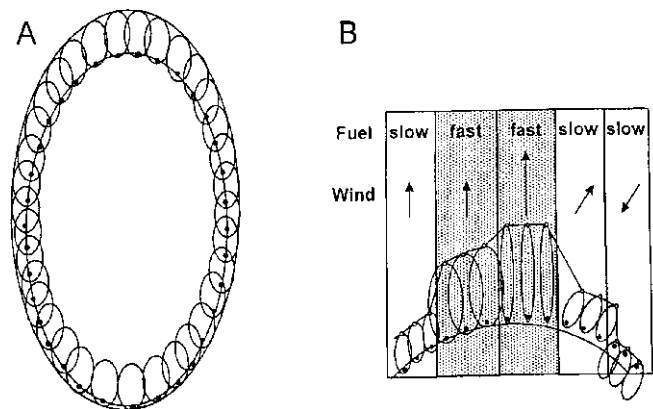


Figure 5—The fire growth algorithm of FARSITE uses a series of ellipses (Finney 1998). A. Under constant weather and fuels, these “wavelets” are of constant shape and size. B. Non-uniform conditions show the dependency of wavelet size on the local fuel type but wavelet shape and orientation on the local wind-slope vector.

Given accurate input data, the model is consistent with expectations for fire growth of surface fires. Spotting and crown fire spread are not possible to verify, although simulations do produce patterns that resemble phenomena observed on real fires. Outputs for FARSITE are geographically referenced, and flame length or fireline intensity per cell can be exported to fire effects and stand growth models to simulate landscapes over time (for example, Keane and others 1996 a,b). For wilderness applications, FARSITE could be applied to generate behavior under worst-case conditions to evaluate possible escape scenarios over a summer for a prescribed natural fire, and could be linked to ecological effects. If adjacent fuelbreaks are proposed adjacent to wilderness as a rationale for loosening prescriptions for fire within wilderness (Agee 1995), FARSITE can be used to evaluate effectiveness of the fuelbreak (van Wagtenonk 1996) and spatial effects on fire control efficiency (Finney and others, in press).

Few wilderness areas have databases that allow application of FARSITE. Yosemite National Park was on-line early due to the presence of an advanced geographic information system (J. van Wagtenonk, personal communication). Where FARSITE data layers (elevation, aspect, slope, fuel model, canopy cover, height to crown base, crown bulk density and canopy height) have been generated, accuracy levels are sometimes so low (Keane and others 1998) that application of the FARSITE model is bound to produce uncertain results, even if weather variables were perfectly predicted.

One of the major lessons learned in the 1988 fires was that the Rothermel fire spread model was not particularly robust in predicting the behavior of fires that contained a large degree of crown fire activity (Thomas 1989). Most of the quantification of conditions where crown fire occurred was derived from boreal forests of Canada (Van Wagner 1977). Crown fire assessments were possible (Alexander 1988) but not routinely employed by wilderness fire managers. After the 1988 fire season, it was apparent that better understanding of crown fire behavior was needed. Rothermel (1991) evaluated crown fire potential in northern Rocky Mountain forests, and his derivation of crown fire spread was empirically derived as 3.34 times the surface fire rate of spread of NFFL fuel model 10. Links of forest structure (Agee 1996) and weather conditions (Scott and Reinhardt, in press), using the Van Wagner and/or Rothermel approaches, have been made and are incorporated into the landscape model FARSITE (Finney 1998). Nevertheless, all involved in this research recognize the imperfect level of our understanding, and the difficulty of experimentation with crown fire only slows progress.

Monitoring

One of the deficiencies of wilderness fire programs in the early 1980s was inadequate monitoring of the fires. Most programs did have monitoring programs “on the books,” but funding was often inadequate. Close monitoring occurred on early prescribed natural fires (for example, Daniels 1974), but as programs expanded, research and monitoring activities became a bit more haphazard. No standards existed for how or what to monitor or how intensive monitoring should be. Some programs had few records of where they had used prescribed fire or what prescriptions were applied (Swezy and

Agee 1991), as some programs were satisfied that fire was “back on the land” and effects were therefore natural. The review of visual effects of fire in giant sequoia groves led to a recommendation that a formal monitoring program be implemented for all national park fire management programs (Christensen and others 1987). This led to the development of the Western Region Fire Monitoring Handbook (NPS 1990), which was widely adopted through the NPS fire programs when funding mechanisms changed (Botti and Nichols 1995). Fire monitoring teams are base-funded, using emergency pre-suppression dollars, and supplemented with additional personnel during periods of high fire activity.

Levels of monitoring activity are defined, recognizing that not every fire requires the same degree of monitoring. Parks that cannot comply with the guidelines do not have a fire program. Level 1 covers reporting of all fires, and levels 2, 3, and 4 call for monitoring of fire conditions, short-term effects and long-term change, respectively. The levels are cumulative, so that requirements for one level include all those above it. Monitoring at all four levels is required for prescribed fires, while prescribed natural fires may include levels through 2, 3, or 4. While the monitoring is not research, meta-analysis of these data might be so considered, and the monitoring may suggest hypotheses that can be experimentally tested (NPS 1990).

This monitoring protocol has been widely adopted in the National Park Service and other state and federal agencies (Reeburg 1995). Training courses have been developed and implemented, and periodic refinements are expected.

Meeting the Challenges of 1986

The last state-of-knowledge review (Kilgore 1986) defined directions for future research in two broad categories: techniques/methods research and new information needs. I have chosen to rate our progress subjectively in those areas, ranking both the quality of the question and the degree of progress we have made since then.

Techniques/Methods Research

1. *Develop criteria by which managers can judge whether an ecosystem has been impacted in a major way by past fire suppression/exclusion.* Kilgore (1986) suggested that both fuels and forest structure must be addressed. We have made significant strides in this area, but the technology was available for fuels well before 1986 (Van Wagner 1977). For low-severity fire regimes, criteria for estimating surface fire intensity (Albini 1976; Burgan and Rothermel 1984; Rothermel 1972), torching potential and crown fire spread potential (Van Wagner 1977) have suggested that many ecosystems are capable of severe fire behavior where that behavior was once rare (Agee 1996). For high-severity fire regimes, the introduction of the concept of nonequilibrium systems has so broadened the sets of possible ecosystem states that the impact of fire exclusion, where fire return interval was historically >100 years, has become fuzzier.

2. *Develop minimum impact methods for determining fire history in wilderness and park ecosystems.* A good fire history technique requires a minimum impact method, but it still may be intrusive to wilderness character. In low-severity

fire regimes, there is no substitute for wedge samples that record tree-ring widths and scars. In high-severity fire regimes, stand ages can be sampled with little visual impact, whether one uses the natural fire rotation, negative exponential or Weibull methods (Heinselman 1973, Johnson and Van Wagner 1985). It appears unlikely that correlation of broad descriptors such as forest type will be sufficient to predict the central tendency and variation in fire regime in a given park or wilderness.

3. *Develop cost-effective techniques for restoring natural conditions over extensive areas of a wilderness or national park and demonstrate these methods.* We have made no progress in this area, as it is largely a management-oriented question, and area burned has declined so much that little information would even be available to analyze.

4. *Develop standard techniques to help managers monitor performance of their wilderness fire plans.* The NPS (1990) monitoring plan has been largely successful in providing a basic outline for monitoring requirements for both prescribed fires and prescribed natural fires.

5. *Develop the capability to predict August behavior of natural fires ignited in July in wilderness areas.* FARSITE has given us the technical capability to provide this prediction capability, but it requires very precise, short-term weather data that are lacking in most wilderness areas.

6. *Develop special techniques for simulating the natural role of fire in wilderness areas where allowing natural (lightning-caused) fires to burn is impractical and where ignitions outside the wilderness no longer burn into the wilderness.* We have made no progress in this area in the United States; in Canada, this technique is still controversial due to the uncertainties expressed in the first technique discussed above (Weir and others 1995, Weirchowski and others 1995).

New Information Needs

1-3. *Using the best data available, determine the "natural" fire history, fire behavior and fire effects for key short-return-interval wilderness ecosystems. Document with case studies, in key short-return-interval ecosystems, how significantly current conditions depart from "natural" in terms of fuels and forest structure. Decide how precise we must be in restoring fuel levels and forest structure to key short-return-interval ecosystems before we allow natural fires to burn again.* I have combined the first three because they all are related, although not exclusively, to the giant sequoia fire restoration controversy of the mid-1980s. These questions have not been addressed in other ecosystems to the same extent they have in sequoia groves, where there has been a considerable investment in science (Stephenson 1991; Stephenson and others 1991; Stohlgren 1993). The answers have generally been that there are wide boundaries on what is considered natural and that, in the process of restoration, care can be taken to avoid effects that, even though natural and perhaps essential for sequoia reproduction, cause public concern, high bark char being an example (Cotton and McBride 1988).

4. *What is the relative importance of aboriginal ignitions in determining intervals between fires and both intensities and severities of fire?* This question is impossible to answer,

and no effort has been expended in any quantification of an answer.

5. *Determine whether scheduled fairly high-intensity prescribed burns can approximate the ecological effects of high-intensity, stand-replacing fires under less explosive burning conditions.* See #6 under Techniques/Methods.

6. *Determine fire effects relationships to habitat needs of endangered wilderness wildlife species such as the grizzly bear.* An entire conference was devoted to rare and endangered species issues and fire (Greenlee 1997). Although production function relations were largely lacking (such as fire at 'x' level results in 'y' response from wildlife or plants), it was clear that many of these species have some tolerance to fire, and some may be dependent on it.

7. *Determine how suppression of fire has impacted key insect and disease populations in certain forest types.* Both insects and disease may attain outbreak or epidemic conditions where plant vigor is low. Due to factors beyond fire suppression, long-term reconstructions may be needed to tease out the "fire exclusion" effect from natural variation over time (see, for example, Swetnam and others 1995). We have made some progress here, but future progress is needed and likely.

Challenges for the Future

The next wilderness conference may well have state-of-knowledge papers that will critique progress after this conference. I have chosen a more restrictive set of challenges than did Kilgore, and I hope for a higher degree of success, at least through the semantic ruse of having fewer categories.

Fire Island: Wilderness in Linked Landscapes

Natural resources planning has increasingly moved to tiered approaches at various scales to account for species and process issues that are important from broad to fine scale. Park and wilderness areas, because of their relatively unspoiled ecological conditions, can be considered core areas. Often, these areas have the highest ecological integrity of regional landscapes and serve as buffers to managed landscapes (Quigley and others 1996). Conservation strategies based on incorporating or simulating historic disturbance processes are thought to have a high probability of maintaining ecological integrity (Dale and others 1999), and they are the basis of the coarse-filter/fine-filter conservation approach (Hunter 1990). In this approach, quite consistent with wilderness management, the ecological processes, including fire, are allowed to interact as naturally as possible, and they therefore help to maintain the conditions that provided the biological diversity of the ecosystem.

Where this coarse filter fails, then species-specific fine filter plans are implemented. Many wilderness areas are surrounded by managed landscapes where past management has placed species at risk. Those species often have required fine-filter plans to maintain certain vegetation structures on the landscape. A good example is the northern spotted owl in the Pacific Northwest, which favors old-growth forest. Most of the remaining old-growth is in parks

and wilderness, and these areas provide a core of habitat for the owl. Where the old-growth is in a natural high-severity fire regime, fire will destroy owl habitat locally. While this may be consistent with a coarse-filter conservation strategy for wilderness, it may be incompatible with a fine-filter conservation strategy for owls. In spotted owl habitat with low- and moderate-severity fire regimes, lower intensity fires may be necessary for and compatible with maintaining old-growth structure.

There are likely to be increasing numbers of fine-filter plans that may conflict with coarse filter conservation strategies (Agee 1999), and fire will likely be a key issue. In a complex natural resources management environment, fire will have to be judged on both its short- and long-term effects at scales well beyond the wilderness boundary. Wilderness is not an island, but science has not yet comfortably placed wilderness in the ecological context of neighboring landscapes.

Ecology and Behavior

The nature of scientific challenges will differ by fire regime. While both ecological and behavioral issues remain for all fire regimes, the ecological ones appear largely in low- and moderate-severity fire regimes, while the behavioral ones dominate the high-severity fire regimes. What we have learned about fire regimes in the last decade is that they are more complex than previously described, and management plans need to address these complexities. Additional research in the parameters of fire regimes (both central tendencies and distributions) will help fine tune future management planning.

One of the more profound lessons we have learned over the past decade is that patchiness and variability are important ecological determinants of fire effects. For example, simulations such as FIRESUM (Keane and others 1989) show Douglas-fir to be almost absent where the fire-return interval in inland Northwest ecosystems is 10 years. That result is because fire return interval is fixed, and the simulated fire burns every piece of the simulated landscape. Real fires are patchy and variable, and we do find Douglas-fir on these landscapes, although it is subordinate to ponderosa pine. One of the major challenges of the next decade is to realistically incorporate patchiness into our simulation models. At landscape levels, FARSITE (Finney 1998) may be able to accommodate patchiness if the cell size is designed to be sufficiently small. Each cell will still burn as a homogeneous unit, but the variability on the landscape will be more realistically simulated. Perhaps a link to fire weather will allow a scaling of fire coverage by cell: At high fuel moisture and low wind, more patchiness will be allowed; as fire weather becomes more severe, less patchiness will result.

Incorporating synergism into future models will be important. This was tried in rudimentary fashion by earlier models (such as Keane and others 1990), and is in progress in currently developing watershed simulations (K. N. Johnson and J. Sessions, Oregon State University, personal communication). Linking fire effects to those of wind, insects and disease will be important to realistic ecological models of the future. It is possible to conceive a realistic model. Wind effects are largely a function of stand structure and topographic location. Insects are target (species)-specific organisms, both

at endemic levels and at epidemic levels once a basal area threshold is exceeded. Organisms causing disease are similarly focused on target species and may be more or less important in various potential vegetation types.

Fire behavior models for wilderness are now far ahead of the databases available to test the models. Continued work on refining methods to collect accurate GIS layers will be necessary. The models will need better criteria for transition from surface to crown fires, and better ways of modeling crown fire behavior. Uncertainty will always remain, due to unpredictable events such as low-level jet stream movement or the movement of plume-driven fires, but the envelope of uncertainty can be significantly narrowed from where it is today. None of these models will work well without good weather information.

Fire Weather and Climate

The accuracy of fire behavior models is highly dependent on good fire weather information. Recent fire model applications (Keane and others 1996a) continue to note the lack of good weather information for wilderness. In some areas, there is no local information at all. Our future requirements are not only for longer-term local weather, but for very specific parameters on hourly time steps, if we want to accurately predict future fire (within limits, of course) or even reconstruct historic events (Cohen 1991). The network of fire weather stations where long-term fire weather data are collected is largely outside of park and wilderness areas, so extrapolation of these data to local conditions is necessary. This limits the ability to simulate future activity of currently active fires or gaming of possible future fires. Expansion of fire weather stations within wilderness, technically feasible now with RAWS (remote automated weather stations), is needed for both research and management purposes.

The existing ability to project climate change, due to either natural change or global warming from human activities, is poor. Effects on distribution of vegetation and possible drivers of that change, such as fire are also largely unknown and speculative. Better models will start with better climate projections, which now appear to deal with temperature much better than moisture, and even then are not very reliable at subregional scales. Current projections of increases in area burned (such as Flannigan and Van Wagner 1991; Romme and Turner 1991) are largely speculative. Global warming may increase fire activity, but coastal evidence suggests major fires during global cooling episodes (for example, Agee 1993). The primary research need is better climate scenarios, followed by research on effects of such climate shifts on structure (vegetation) and process (disturbance, broadly defined).

The Need for Courage

Natural resources science often does not provide specific answers to operational problems. At best, it may provide limits or boundaries on uncertainty, or it may increase the uncertainty of the manager's domain. This may be very pleasing to a scientist, but it may leave the manager with a

longer list of what might go wrong. In wilderness fire science, the political triggers are much more oriented to fire behavior than to fire effects. The consequences of the long-term effects of fire exclusion, or the severity of an individual fire, are much less likely to be on the manager's radar screen than a fire that escapes a wilderness boundary. While the scientific community has made progress in both the ecological and behavioral domains of wilderness fire, we have still a long way to go. Ironically, one of the important ways we can learn from wilderness fire is to do it and accept the uncertainty in the process. Continued progress can occur in the laboratory and in the computer, but the land is where wilderness fire science must be evaluated. Wilderness fire managers face a real challenge, as even the most successful "people managers" will always be failures at managing the weather. There always will be subtle pressures to avoid a commitment to wilderness fire programs. Successful wilderness fire management will require continued generations of courageous managers (Daniels 1991, Kilgore and Nichols 1995). The success of wilderness fire science depends on it.

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