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Review

Research and development supporting risk-based wildfire effects prediction for fuels and fire management: status and needs

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Abstract. Wildland fire management has moved beyond a singular focus on suppression, calling for wildfire management for ecological benefit where no critical human assets are at risk. Processes causing direct effects and indirect, long-term ecosystem changes are complex and multidimensional. Robust risk-assessment tools are required that account for highly variable effects on multiple values-at-risk and balance competing objectives, to support decision making. Providing wildland fire managers with risk-analysis tools requires a broad scientific foundation in fire behaviour and effects prediction as well as high quality computer-based tools and associated databases. We outline a wildfire risk-assessment approach, highlight recent developments in fire effects science and associated research needs, and recommend developing a comprehensive plan for integrated advances in wildfire occurrence, behaviour and effects research leading to improved decision support tools for wildland fire managers. We find that the current state of development in fire behaviour and effects science imposes severe limits on the development of risk-assessment technology. In turn, the development of technology has been largely disconnected from the research enterprise, resulting in a confusing array of *ad hoc* tools that only partially meet decision-support needs for fuel and fire management. We make the case for defining a common risk-based analytic framework for fire-effects assessment across the range of fire-management activities and developing a research function to support the framework.

Additional keywords: decision support, integrated assessment, spatial scale, temporal scale.

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Introduction

Fire planning and management strategies that consider the role of wildfire and fire effects, including using wildfire to restore fire-adapted systems (Dickinson and Ryan 2010), contribute to ecosystem health (Keane *et al.* 2008). However, considerable uncertainty regarding the ecological effects of wildfire (Thompson and Calkin 2011) limits the potential to improve ecological conditions through wildfire management (Calkin

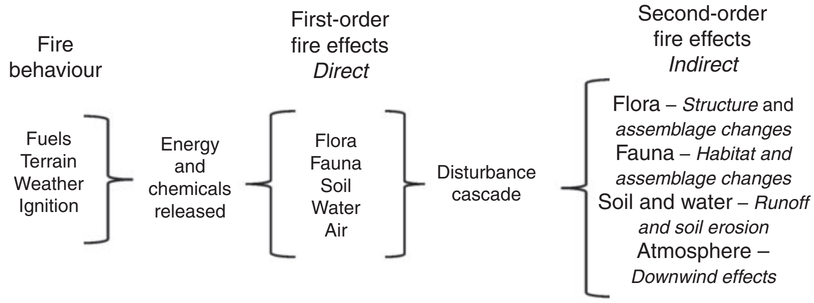

Fig. 1. Relationship between fire behaviour and first- and second-order fire effects.

et al. 2011*a*). Assessment of potential risks and opportunities associated with wildland fire requires analysis of probable fire behaviour and the likelihood of multiple fire effects on natural resources and ecosystem processes and components over a range of probable outcomes (Finney 2005).

Fire effects follow from fire behaviour and may be direct (first order) or indirect (second order) (Reinhardt *et al.* 2001) (Fig. 1). Multiple factors interact to drive fire behaviour and control heat release that directly affects flora, fauna, soil, water and air. Successive temporal and spatial linkages between resources, components and processes shape disturbance cascades of indirect effects (Nakamura *et al.* 2000), which may be delayed over time and extend for long distances from the area burned. Fire effects may be neutral, negative or beneficial, depending on fire intensity, societal values, susceptibility of ecosystem components to fire effects and how fire may influence management options moving forward (Miller and Landres 2004; Keane *et al.* 2008).

Difficulties with prediction of fire effects arise because of gaps in core fire and fire-effects science, limited transfer of existing fire-effects knowledge, and spatially inconsistent and limited databases needed to support analysis. Prediction is complicated and inherently uncertain because of the chaotic, multi-scale and non-linear nature of the physical processes that govern weather, fire and ecosystems (Peters *et al.* 2004). Biotic and abiotic ecosystem factors interact and adjust through spatial and temporal webs that are often complex and difficult to understand and characterise (Briske *et al.* 2005; Bowman *et al.* 2009). Further, assessment of fire effects necessitates trade-off analysis. Fire effects may lead to benefits for one resource of concern while harming another (Boerner *et al.* 2006; Sugihara *et al.* 2006). Short-term losses may be tolerable in exchange for longer term gain whereas some losses may be intolerable in any time frame.

The complexity of fire-effects analysis and need for decision support has driven development of multiple software systems designed to characterise, assess, and simulate fire behaviour and effects for management application. The confusing array of fire behaviour and effects modelling tools and gaps in core fire-effects science reflects *ad hoc*, disconnected research and development programs (Palmquist 2008). We propose that improvement in fire-effects prediction capabilities begins with defining a common risk-based analytic framework for fire-effects assessment across the range of fire-management activities. A consistent framework serves as a prerequisite for improving fire-effects science and further development of decision-support software tools (D'Erchia *et al.* 2001) and assures that common methods are employed to plan, prioritise and document environmental management actions based on best available science.

The purpose of this paper is to make the case for defining a common risk-based analytic framework for fire-effects assessment across the range of fire-management activities. We propose developing a research function to support the framework. Three sections follow: first, we define risk-based assessment and analysis of fire effects; second, we survey the status of fire behaviour and effects science relative to prediction needs; and finally, we summarise gaps and present a general framework for risk-based fire-effects analysis.

Risk analysis in wildfire management

Wildfire risk depends on the probabilities of fire behaviour and fire effects (Finney 2005). The idea of a wildfire risk-management framework implies a systematic and repeatable assessment process that evaluates the probable success and effects of proposed actions to meet multiple, commonly competing resource-management objectives. Prior discussions of wildfire risk analysis include a wildfire risk-assessment framework built around a core risk matrix that accounts for probable ignition, fire behaviour and fire effects (Schöning *et al.* 1997; Bachmann and Allgower 1998); introduction of a theoretical risk framework based on core concepts of likelihood or probability, consequence or effect, and objective or basis for measuring consequence (Shields and Tolhurst 2003); a comprehensive approach to wildfire risk management based on principles of environmental risk assessment (Fairbrother and Turnley 2005); and a demonstration at broad spatial scales of risk to valued resources based upon burn probability and expected loss (Calkin *et al.* 2010).

For our purposes we adopt a definition of wildfire risk quantified as: the expected net value change to resources due to fire of given energy-release characteristics or intensity, times the probability of occurrence of a fire at that intensity, summed over all possible fire intensities (Finney 2005). Fire intensity (kW m^{-1}) is not the only relevant determinant of fire effects (Alexander and Cruz 2012) but we use it as an example. Fig. 2 presents a sequence for risk-based analysis that explicitly accounts for fire effects in wildfire decision support and provides a basis for building a risk-based procedure for assessing fire effects.

Assessment of wildfire risk adjusts potential losses or gains (whether quantified monetarily, by simple counts of acres or population affected, or other means) for the likelihood that

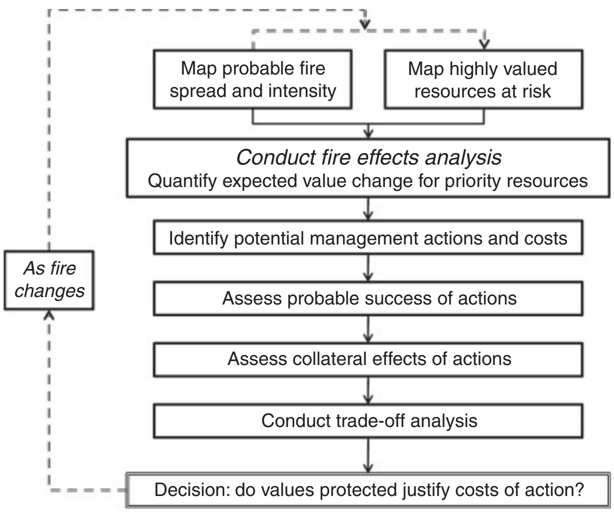

Fig. 2. General framework for assessing wildfire risk to support management actions (After Finney 2005; Calkin *et al.* 2007 and Thompson *et al.* 2011).

valued resources will be exposed to fire and the probability that effects will occur. The probability that a fire once ignited will burn a given area is dependent primarily on fire weather conditions, antecedent climate and the condition of available fuels. Change in resource value depends on how a given resource responds to fire outputs, i.e. energy and chemical releases. With risk quantified, an objectively described judgement follows to qualify the significance of potential loss or gain due to fire effects.

A major impediment to implementing risk-based effects analysis is lack of ability to quantify or qualify expected value change for valued resources (Calkin *et al.* 2011*a*) (step 2 in Fig. 2). Historically, value change analysis has been done well only where private property was threatened and a market value could readily be assigned. Loss was commonly expected with the unquestioned assumptions that first a fire would reach, for example, a structure and second, that the structure would suffer total loss from fire. Quantification of value change for non-market resources – natural resources and ecosystem services – is limited due to gaps in fire-effects knowledge (Calkin *et al.* 2011*a*), difficult access to the knowledge that has been published (J. K. Smith, USDA Forest Service, pers. comm.) and misunderstanding about valuation of non-market resources (Shields and Tolhurst 2003; Calkin *et al.* 2008).

If risk analysis predicts probable and significant negative consequences if no action is taken then management actions that might change outcomes may be considered. Another probability assessment is made, this time to assess the likelihood that proposed actions will be successful. Next, potential unintended or collateral side effects of proposed actions enter into consideration. Following that, managers use trade-off analysis to evaluate relative success of proposed actions in bringing about desired outcomes for the range of resources-at-risk. In the final step managers judge whether expected resource losses justify the costs of proposed actions.

This general risk-assessment process may be applied to the fuel- and fire-management continuum (Fig. 3). Activities and

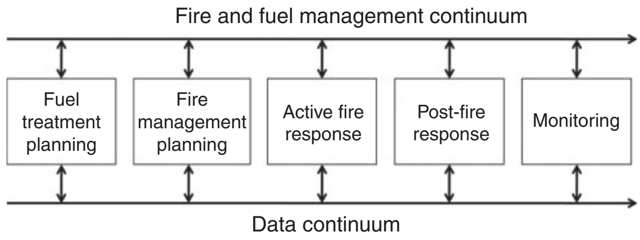

Fig. 3. Fuel and fire management and data continuums with cumulative and interactive activities and data flows.

data flows are cumulative and interactive. Each phase requires fundamentally identical data about biotic and abiotic conditions to characterise fire behaviour and effects. These linkages provide compelling support for developing seamless decision-support systems to conduct assessments, plan activities, monitor results and systematically store and efficiently retrieve all required data. Differences between spatial and temporal scales constitute the primary differences between management phases. Fundamental relationships between fire behaviour and fire effects remain constant as does the need for spatial inventories of values threatened by wildfire. Advances in core fire-behaviour and effects science and their supporting datasets are required to support development of effective, integrated decision-support technology. Likewise, consistent, up-to-date fuels and vegetation data are required to drive fire-behaviour and effects predictions. Post-fire inventory and monitoring data are needed to update fuels and vegetation data and evaluate the action's outcome, for example, success *v.* failure.

Status of fire-behaviour and fire-effects science and its scaling in time and space

Fire-behaviour measurements or predictions from models are required to predict fire effects (Fig. 1) and, thus, advances in fire-effects science are dependent on the development of fire-behaviour science. Although existing knowledge can be better deployed in decision-support technology, for example, through improved integration and better foundational datasets, gaps in core fire-behaviour and effects science will ultimately limit ecological-effects prediction and its use in these systems. In the following, we focus on limitations in both fire-behaviour and effects science and address challenges of spatial and temporal integration.

Fire behaviour

Fire-effects prediction would benefit from a range of improvements in fire-behaviour quantification and modelling starting with a focus on providing fire-model outputs needed for effects prediction, not just for fire operations. Benefits will also accrue from improved fire-model validation and calibration, continued development on the frontiers of fire modelling and advances in provisioning the input data (e.g. fuels and weather) required by fire models. Finally, advances in fire- and fuels-measurement methods will provide new sources of input data, especially spatial, for effects prediction. These issues have largely been reviewed in previous work and we will only summarise them here. Detailed discussion of fire-behaviour prediction is beyond the scope of this paper; the reader is referred to recent reviews (e.g. Sullivan 2009*a*).

Fire-effects prediction is hindered by missing information, requiring not only fireline intensity and rate of spread that are so important for fire operations, but also characterisation of the total and time course of heat and smoke release (Dickinson and Ryan 2010). For instance, a flame model and information on fireline intensity, rate of spread and flame residence time could be used to predict the time course of tree stem heating (Butler and Dickinson 2010), but flame characteristics and residence time must currently be estimated from *post hoc* engineering correlations (Bova and Dickinson 2008). Albini and Reinhardt (1997) developed the most promising approach to predicting the total and the time course of heat release. Their process model estimates rate of burning of large-diameter woody fuels and total consumption and requires fuelbed and fuel-moisture information as inputs. The model functions as the core source of fire characteristics in an operational model that simulates first-order fire effects (Reinhardt 2003) but requires evaluation in a wider range of ecosystems and explicit extension to other fuel components. Another fuel-consumption prediction system (Prichard *et al.* 2010) also has the potential to contribute to effects prediction because it provides fuel-consumption predictions, which are directly proportional to total heat release (Kremens *et al.* 2010).

Considerable work is needed on predictive modelling of smouldering combustion, the independent flameless combustion of organic soils horizons (Rein 2009) and large woody material (de Souza Costa and Sandberg 2004). Smouldering duff can cause mortality of large, old trees in ecosystems that developed under a regime of frequent fire but where fires have long been absent (Ryan and Frandsen 1991; Varner *et al.* 2005). Duff consumption is a factor in soil heating and is often a major source of pollutants from wildland fires. Duff loss, either from combustion during flame front passage or by subsequent smouldering, is a primary determinant of post-fire erosion (Robichaud 2000). Duff smouldering is a function of duff depth, moisture and other factors (Miyanishi and Johnson 2002; Varner *et al.* 2007) and its prediction depends on fine-scale duff moisture prediction and fuel type. Duff moisture and depth often vary at the scale of individual trees because of litter deposition and rain interception patterns (Miyanishi and Johnson 2002). Despite its importance and past and ongoing research, no operational smouldering model is available.

A poor state of fire-behaviour model evaluation exists, which is not only problematic for fire operations, but also for fire-effects prediction. Given the near ubiquity of the core Rothermel (1972) algorithms in operational fire-behaviour prediction in the United States, it is useful to note that the model has not undergone significant modification since publication despite the passage of 40 years and subsequent experimental work intended for its improvement or replacement (Catchpole *et al.* 1998; Finney *et al.* 2010). Also, field calibration of the Rothermel model has not been conducted in a formal or organised way (Sullivan 2009*b*) despite known biases (Grabner *et al.* 2001). Similarly, no validation of the fuel-consumption prediction system in Prichard Ottmar *et al.* (2010) has been published to date though a validation project is in progress and work continues on estimating model parameters for unrepresented ecosystems (e.g. eastern mixed-oak forests). More positively, the large woody fuels combustion model has undergone calibration and validation using data from natural and slash-dominated fuelbeds in forested areas of the north-western US (Albini and Reinhardt 1997), though Reinhardt's (2003) first-order fire-effects model, of which the work of Albini and Reinhardt (1997) is a component, is applied in many ecosystems for which no model evaluation has been conducted (e.g. non-forest fuels). Also, although the core fire behaviour algorithm (Rothermel 1972) has been poorly evaluated and calibrated, a landscape implementation used during active fire operations in probabilistic mode has undergone validation to assess overall accuracy (Finney *et al.* 2011). Initial results of current work at basic levels of fire spread and combustion dynamics (e.g. Finney *et al.* 2010) support fundamental revisions of fire-behaviour theory, which could produce more accurate predictions.

Coupled fire–atmosphere (CFA) models simulate feedbacks between the fire and the atmosphere and can generate more realistic fire behaviour than can non-coupled models (Linn and Cunningham 2005; Mell *et al.* 2007; Parsons *et al.* 2011). First-order fire-effects prediction will benefit from CFA modelling because, to varying extents and by different means, they explicitly predict heat dissipation that can be used to provide the input data needed by process-based fire-effects models. However, substantial work will be required to couple CFA outputs to fire effects. Coen (2005) demonstrated better-than-real-time predictions using a model that couples Rothermel's empirical approach and the large, woody combustion algorithms of Albini and Reinhardt (1997) with an atmospheric model (Clark *et al.* 2004). For support of fire-effects modelling, and as a means of overcoming computational limitations, useful simulations from models more computationally intensive than Coen's (2005) can be conducted at relatively fine spatial scales (Parsons *et al.* 2011) and the results of multiple model runs summarised in look-up tables or functional relationships for real-time application (e.g. Bova *et al.* 2011).

Improvements in fire behaviour predictions will accrue from improvements in the accuracy and spatial (and temporal in some cases) resolution of input data. Improvements in meteorological inputs would appear to be a low-hanging fruit. As has been shown by Potter and Butler (2009), including high-resolution wind fields in simulation models would considerably improve fire-behaviour predictions. Better fuel-moisture predictions would also improve the reliability of fire-behaviour predictions. Currently, only woody and live fuel-moisture models are operational in the US. There is no fuel-moisture model for litter and duff (i.e. layered fuels), a major gap in capabilities. Operational woody fuel-moisture models are simplistic and generally focussed on worst-case scenarios, which may not be appropriate for effects prediction. Physically based woody fuel (Nelson 2000) and layered fuel-moisture models (Matthews *et al.* 2007) are available for implementation yet require inputs not available from operational fire-behaviour systems.

Dynamic global vegetation models (DGVMs) have the potential to provide spatial inputs for process-based fuel moisture models. Although implemented primarily as research tools (e.g. Medvigy *et al.* 2009; Sato *et al.* 2007), DGVMs have the potential to contribute substantially to the quality of input information for fire and fire-effects modelling because of their more realistic representation of vegetation. Despite their name, DGVMs use input from atmospheric models at different resolutions, varying (in different models and different simulations) from global and regional to local resolution, and predict the interaction between the atmosphere and the ecosystem.

Developments in fire and fuels measurement (*sensu* metrology, see review in Kremens *et al.* 2010) bode well for future advance in fire-behaviour and effects prediction. Use of airborne light-detecting and ranging (LiDAR) technology to characterise pre- and post-fire fuelbeds at fine resolution over large areas and relating those fuels to behaviour is an active area of research with considerable promise (Skowronski *et al.* 2011). Ground-based LiDAR has also shown promise in characterising surface fuels at high resolution (Hiers *et al.* 2009). Airborne and ground-based LiDAR have been combined to increase map resolution (Skowronski *et al.* 2011). Remote (airborne and fixed) fire-radiation mapping has been applied to fire-behaviour quantification (Radtke *et al.* 2000) as well as ecological problems (Riggan *et al.* 2004). Ground-based instruments, placed so as to be burned over or underneath in fires are under continued development and will lead to improved datasets for fire- and smoke-model evaluation (Clements *et al.* 2007; Kremens *et al.* 2012).

First-order fire effects

A range of science problems require solution before reliable predictions of first-order fire effects can be made in the context of fuels and wildfire-management decision support. A recent Fire Ecology special issue focuses on strengthening the science basis for first-order fire-effects prediction (Dickinson and Ryan 2010). General areas that require focus include drawing linkages between fire-behaviour modelling outputs and fire effects and the development of effects models themselves. Where tools for predicting effects from behaviour exist, model evaluation is often lacking (though, see Hood *et al.* 2007).

Advance is needed in the various means (e.g. statistical or process based) of linking fire behaviour with effects on soils, vegetation and fauna. Currently, the linkages between fire behaviour and effects are underdeveloped (i.e. non-existent, simplistic or untested). Process modelling relates fire energy dissipation to surface heating of soils and vegetation. With surface heating information, heat- and mass-transfer models are used to describe effects on soil or vegetation. A model produced by Campbell *et al.* (1995) provides soil-surface heat fluxes for the soil-heating model in Reinhardt's (2003) first-order fire-effects model, but this linkage has not been peer reviewed or, apparently, validated and cannot account for the insulating effects of unburned duff (Massman *et al.* 2010). Relative to effects on vegetation, there are no validated approaches to linking surface-fire behaviour with tree stem heating (which is generally uneven around a stem) although both statistical and more process-based approaches have been proposed (Butler and Dickinson 2010). Michaletz and Johnson (2008) and Kavanagh *et al.* (2010) use a plume model to provide boundary conditions for branch injury but no validation has been conducted. As a means of characterising faunal exposure to smoke, Bova *et al.* (2011) use a computational fluid dynamics model to link plume characteristics with exposures to fauna inside tree cavities and, again, field validation is needed. Statistical relationships between fire behaviour and effects based on field experiments exist including Van Wagner's (1973) crown-scorch model and Bova and Dickinson's (2005) tree stem-necrosis relationships, but how generally valid they are is not known, though Van Wagner's relationship is widely used.

Better understanding and continued development and testing of process-based fire-effects models themselves is also needed in concert with the development of linkages between fire and effects models as discussed above. Fire-effects prediction in herbaceous and shrub vegetation has lagged behind work on trees. The first model of fire-caused rhizome bud death in grasses has been published (Choczynska and Johnson 2009) and should be evaluated. We know of no model describing soil-heating effects on soil seed banks (see Stephan *et al.* 2010). Indeed, many fire-effects problems are related to soil heating and, though a first-order model of soil heating exists (Campbell *et al.* 1995), it may not capture processes relevant to intense burning. For instance, previously unknown pressure-induced gas flows in soils during pile burning have recently been proposed to explain massive spikes in soil CO_2 concentrations during fires (Massman *et al.* 2010). Despite a large literature on tree injury and mortality (e.g. Butler and Dickinson 2010; Hood 2010), no validated tree mortality model exists that integrates injury to roots, stems and crowns, though one has been proposed (Michaletz and Johnson 2008). New perspectives on the physiological basis for tree injury continue to emerge, for instance: the relationship between growth efficiency and mortality of large, old trees (Kolb *et al.* 2007); the central role of plume vapour pressure deficits in causing crown injury (Kavanagh *et al.* 2010); and the role of fine root loss from duff consumption in post-fire stress (O'Brien *et al.* 2010). Extensive theoretical, laboratory and field work is required in a range of areas and, for existing process models, further development (often extensive) and validation are needed. Advance in first-order fire-effects prediction will support second-order effects prediction (Fig. 1).

Second-order effects

Second-order fire effects include interacting (coupled) changes in hydrology, sediment flux, biogeochemical cycling, and changes in vegetative composition and structure and faunal habitat that result from individual fires and fire regimes. We survey recent trends in second-order effects science related to coupled land-surface processes and faunal habitat. Detailed discussion of science gaps for broad topic areas are found in the 'Rainbow Series' publications including fire effects on flora (Brown and Smith 2000), soil and water (Neary *et al.* 2005), fauna (Smith *et al.* 2000), invasive species (Zouhar *et al.* 2008) and cultural resources (Ryan *et al.* 2012). Fire effects on air quality are addressed elsewhere in this issue (Goodrick *et al.* 2012).

Coupled land-surface processes

Modelling and assessment of second-order fire effects related to soil and water depends both on adequate representation of the immediate responses to fires (such as increased runoff, erosion and nutrient losses) and the rate of recovery of these processes during the post-fire period. Core fire hydrology science and integration of post-fire hydrologic and erosion response into applied models is relatively advanced compared with other second-order fire effects. This is likely due to the threat from floods and sediment movement to human life, property, and drinking water supplies that commonly follow wildfires and extensive work with sensitive and endangered aquatic species where fire is one of many disturbance processes of concern (Rieman and Clayton 1997; Gresswell 1999).

The initial hydrologic response will vary both with severity and spatial extent of the fire and a variety of site characteristics, such as topography, pre-fire vegetation, soil type and climate. For example streamflow typically increases during the first year following the removal of vegetation (through fire, insect epidemic or logging). The magnitude of increase depends on vegetation effects – degree of consumption, area of vegetation loss, and damage of regeneration potential (Ryan 2002; Neary *et al.* 2005; Hyde *et al.* 2007); site characteristics including climate, pre-fire vegetation type, geology and topography (Andréassian 2004; Brown *et al.* 2005); and specifically for fires, linkages between fire and soil effects (Cerda and Robichaud 2009).

The importance of pre-fire site characteristics on the sensitivity of streamflow to vegetation loss has been well established and summarised in reviews of the general relationship between vegetation loss and hydrology (Tucker and Bras 1999; Wilcox 2002; Valentin *et al.* 2005). Runoff and erosion response following fire may be more closely coupled to vegetation change than fire effects on soils (Doerr *et al.* 2009; Larsen *et al.* 2009). Further work is needed to clarify the relationship between vegetation change and post-fire erosion. Satellite imagery captures fire effects on vegetation (Hudak *et al.* 2007; Chafer 2008) and burn severity patterns (Chuvieco 1999; Collins *et al.* 2007) and has been used to explain and predict the occurrence of post-fire debris flows (Hyde *et al.* 2007; Gartner *et al.* 2008). The information in satellite images and its relationship to post-fire processes is poorly understood and needs to be further developed. Watershed hydrology models (e.g. Jetten 2002; Tucker *et al.* 2001; Wigmosta *et al.* 1994) capture many site-specific conditions influencing runoff and erosion response. They typically include a vegetation parameter (such as leaf area index (LAI)) that can be varied to estimate the immediate effect of fire on hydrology associated with vegetation loss. Fewer of these models include other fire specific effects (as opposed to general vegetation loss) such as loss of soil organic layers (i.e. duff) and changes in soil texture and infiltration rates.

Estimation of changes in sediment flux, biogeochemical cycling, and nutrient and constituent export in the first year following fire show similar sensitivities to biogeoclimatic setting and burn characteristics (extent and severity) but field-based studies show a wider range of variability and the specific mechanisms are less well understood. Models of post-fire sediment flux also range from largely empirical, such as those based on Wischmeier and Smith (1978), to process based models that consider changes in both surface erosion and transport and shallow landslides after fire (e.g. Benda and Dunne 1997) and explicitly account for post-fire changes in hydrophobicity (e.g. Gabet and Dunne 2003). Robichaud *et al.* (2007) apply ensemble runs of a process-based water-erosion prediction model (Covert *et al.* 2005) to estimate post-fire erosion probabilities from forested catchments and to assess treatment options. The multiple runs in this approach to hydrologic modelling represent a range of probable erosion responses based on climate history thereby building probability statements required for risk-based assessments. These models, however, have received limited evaluation against field data and there remains a need for further development, parameterisation and assessment across a wider range of site conditions.

Biogeochemical cycling, sediment and hydrologic fluxes in the first year following fire are also highly dependent on rainfall and weather conditions (Cannon *et al.* 2008; Sheridan *et al.* 2007). For many regions with high inter-annual variation in climate, the probability of a convective storm event of a given magnitude in the first year following fire is the primary control on the magnitude of response in areas with severe fire effects (Cannon *et al.* 2008). The size of zones of high sediment flux is, thus, often limited to the characteristic aerial extent of intense precipitation within convective storms (Luce 2006) and projections of post-fire effects depend strongly on assumptions of climate patterns and potentially climate change. Uncertainties in prediction of rainfall events that trigger severe runoff and erosion limits accuracy of post-fire erosion predictions (Cannon *et al.* 2008) and may be reduced with refined application of weather RADAR technology (Di Luzio and Arnold 2004; Underwood and Schultz 2004). However, as promising as these approaches are, much work remains to resolve uncertainties regarding accuracy of weather RADAR estimates of rainfall distribution and intensity, especially in mountain environments (Smith *et al.* 1996; Young *et al.* 1999).

The second-order effects of fire typically diminish as vegetation recovers (Lavee *et al.* 1995; Lentile *et al.* 2007). Thus, to adequately capture second-order effects on hydrology, biogeochemical cycling and sediment flux for the longer post-fire recovery period, models must also represent post-fire regrowth. For example, erosion from tree topple after stand-replacement fire in mountain ecosystems declines exponentially after fire but is also a function of tree size and, thus, time since the last fire (Gallaway *et al.* 2009). A variety of models of vegetation growth exist ranging from statistical (e.g. Mason and Dzierzon 2006) to more mechanistic process-based DGVM approaches (e.g. Kittel *et al.* 1995). The family of models discussed earlier to estimate the evolution of fuels are also used to model vegetation recovery (e.g. Reinhardt *et al.* 2003; Keane and Karau 2010). Fewer of these vegetation models are directly coupled with a full hydrology, biogeochemical cycling or sediment-flux model although there are exceptions. For example, models developed by Tague and Band (2004) and Peng *et al.* (2002) couple vegetation regrowth with biogeochemical cycling and hydrology.

In coupling forest-growth models with models of effects, there is often time- and spatial-scale mismatch. Vegetation-growth models can be run using time resolutions – from seasons to centuries – whereas hydrologic–sediment–nutrient cycling processes, particularly for capturing post-fire runoff behaviour, must be run using daily or even sub-daily time steps. Further work is needed to develop coupled sediment, biogeochemical cycling and hydrology models with different post-fire regrowth approaches. Simulation systems that integrate multiple models with greater or lesser degree of redundancy in purpose and nesting in spatial and temporal scale point the way towards a prospective future for second-order fire-effects prediction (Tague and Band 2004; Walko and Avissar 2008; Medvigy *et al.* 2009).

Long-term changes in plant species composition following fire, including increased opportunities for invasive species, can have cascading effects on hydrology, sediment and biogeochemical cycling and these effects do not necessarily diminish with post-fire recovery if they result in broad-scale soil erosion or species change. For example, Gabet and Dunne (2003) argue that sediment yields increase by 40% with a conversion from sage scrub to grasslands, a species conversion that can be facilitated by frequent fire. Although post-fire species distributions can be prescribed in a variety of coupled vegetation growth, hydrology and biogeochemical cycling models, explicit modelling of post-fire species change has not been fully implemented.

Faunal habitat

Fire effects on fauna can be either direct, in the form of modifying behaviour or causing injury or mortality, or indirect through habitat change (Engstrom 2010). Predicting fire effects on faunal populations (second-order effects) requires knowledge, at minimum, of relationships between faunal population status and quantitative features of their environment, as captured in habitat-suitability models (Baird *et al.* 1994; Hirzel and Le Lay 2008). Although much effort has gone into developing habitat-suitability models, they tend to be statistical, not process based, and unverified (Wiens and Milne 1989). A counter example would include a recently published and extensive set of validated habitat-suitability indices for land birds based, in part, on forest inventory and analysis data (Tirpak *et al.* 2006). Given a suitable habitat model that could be applied over a sufficiently ample spatial and temporal domain, it might then be possible to assess whether effects of a given fire or fire regime were acceptable from the perspective of either an individual species or, because a given fire may benefit some species' populations and harm others, a community of species (e.g. Saab and Powell 2005).

We are aware of no published studies in which one of the various fire-effects simulation models or systems have been used to directly predict the effects of individual fires or collections of fires on faunal populations, though projects on fish and birds with that purpose are in progress (R. E. Keane and V. A. Saab, USDA Forest Service, pers. comm.). Faunal habitat models would have to provide targets for modelling that are predictable from fire-effects simulations (Engstrom 2010). Candidate targets include snag availability, vegetation structure, food supply and aquatic system characteristics and their dynamics. Despite the importance of snag abundances, sizes, and species composition for many species of birds, mammals, and insects, snag population models are few and poorly validated. For instance, algorithms for snag dynamics relative to fire effects in Reinhardt *et al.* (2003) are based on unpublished data. One of the few published (mathematical) snag dynamics models is that of Morrison and Raphael (1993), which has limited geographic scope and has not been expanded to new species and ecosystems. The landscape-level fire and succession model by Keane *et al.* (2011) has the capability of simulating effects of fire regimes on wildlife through a habitat-suitability approach in which elements of a matrix of vegetation cover types and age-related structural stages are assigned suitability values for a given wildlife species.

Suitability values might be determined by expert opinion or field research such as has been developed for decision support for use of prescribed fire in fuel-treatment planning. Two such decision-support systems were developed to assess habitat response to prescribed fire. An Australian system (Baird *et al.* 1994) relies on fire response curves developed by wildfire experts that estimate species abundance relative to changes to vegetation structure at different fire intensities. The system accounts for wildlife response as vegetation regenerates over time and as a function of fire-return intervals. A similar approach developed for use in the interior of the western US (Pilliod 2005) includes a broad range of habitat elements for which the user must judge the expected change in each element for a given proposed fuel treatment. This application offers no explicit temporal component. Both systems apply to limited geographic areas and key species. We are not aware of any similar systems developed for use during active wildfire management.

A large body of published works supports the potential for broader development of habitat-suitability applications, much of it collected in the Fire Effects Information Systems (FEIS) (see http://www.fs.fed.us/database/feis/, accessed 9 July 2012). The diverse literature characterises fire effects on species or populations and, with further synthesis, could be used in decision support. However, much of the data are site specific and reflect fine-scales analysis. Prediction of fire effects on fauna and prioritisation of research needs will be severely limited until synthesis is completed and research findings are generalised.

Several promising examples of the use of process models in characterising important habitat elements are available. An individual-tree modelling approach such as used in Keane *et al.* (2011) was applied to the availability of white pine seeds, a key wildlife food in the Rocky Mountains (Keane *et al.* 1990). In an aquatic habitat example, key features of stream flow for Rocky Mountain salmonid populations have been defined and simulated with a hydrologic model (Wenger *et al.* 2010). Channel reorganisation and stream temperature are also known to be important (Dunham *et al.* 2007) and are related to fire effects on vegetation and post-fire erosion along with the coupling between surface and groundwater flow, all of which have been simulated with process models (Buffington and Tonina 2009; Tonina and Buffington 2009). Related to processes governing stream habitats, a millennial-scale erosion model for mountainous terrain showed the greatest sediment-deposition rates in low-order (smaller) streams and illustrated the interactions among weathering rates, gradient and wildfire regime (Martin 2007).

Spatial and temporal integration

Fire is fundamentally a contagion process involving complex interactions among biotic and abiotic factors in time and space (Peters *et al.* 2007) and dependent upon the spatial and temporal dynamics of landscape patches (Turner 1989). To be effective, decision-support tools for wildland fire management must account for the spatial nature of wildland fire phenomena (Blanchi *et al.* 2002; Fairbrother and Turnley 2005; Stockmann *et al.* 2010) Landscape condition and ecological functions result from sequences of multiple, recurrent spatially distributed processes (He 2008) which may exhibit non-linear responses as spatial extent increases (McKenzie *et al.* 1996; Peters *et al.* 2007). Spatial analysis of disturbance processes accounts for interactions within and beyond areas disturbed by wildfire and other processes. Multiple disturbance processes, e.g. those caused by fire, insects, disease, fuel reduction and forestry activities and invasive species, must be accounted for in order to integrate fire and fuels management into comprehensive ecosystem management (Stockmann *et al.* 2010). Forecasting fire behaviour and fire effects for decision support in these systems which exhibit non-linear dynamics (Allen 2007; Davenport *et al.* 1998) requires analysis of cross-scale temporal and spatial interactions if the 'surprises' of catastrophic events are to be minimised (Peters *et al.* 2004).

Existing applications provide partial solutions to spatial and temporal problems. One application (Tirmenstein and Long 2011) works within a geographic information system shell to feed spatially explicit estimates of fire behaviour (Finney 2006) into non-spatial first-order fire effects (Reinhardt 2003) calculations. Multiple spatial units can be modelled limited only by input data and computer capacities. Convenient mapping of outputs facilitates further spatial analysis. No interactions occur between landscape units, each simulation represents a single time period or unique fire event (a static set of inputs leads deterministically to a single outcome) and only fire disturbance is modelled. Another recent and more complex system (Ager *et al.* 2011) extends this fire behaviour to fire-effects logic to model forest stand succession and associated fire behaviour over time in response to management actions and other disturbances. Inclusion of a forest vegetation simulator which specifically addresses fire and fuel conditions (Reinhardt *et al.* 2003) permits simulation of changes in fire hazard in response to different spatial arrangements of fuel-reduction activities and vegetation regrowth. These approaches, reasonably accessible for use by managers, fall short of spatially explicit fire-effects analysis which reflects dynamic interactions between valued resources and ecosystem processes.

Modifications of existing forest-ecosystem research models hold promise of overcoming limitations of fire-effects simulations used by managers for decision support. One example (Keane and Karau 2010) first simulates fire effects from fire behaviour or infers fire effects from satellite imagery (Key and Benson 2006). Application of a landscape succession model (Keane *et al.* 2006) then uses the fire-effects inputs to 'grow' vegetation over 5000+ simulation years and to develop historic range and variability (HRV) time series from which post-fire departure from HRV is predicted. Extensive parameterisation and implementation demands of this application very probably limits operational use for decision support without substantial expert support, a problem common to research models (Perry and Enright 2006). The challenge remains to develop efficient, operational decision-support systems driven by process relationships that integrate fire behaviour and fire effects over multiple temporal and spatial scales and account for multiple, interacting disturbance processes (McKenzie *et al.* 1996; Reinhardt and Dickinson 2010). Foundations of advanced solutions to this challenge may be found in existing general landscape-disturbance models and forest-landscape models (see reviews in Perry and Enright 2006; Scheller and Mladenoff 2007; He 2008).

Framework for risk-based fire-effects analysis

We propose development of a risk-assessment framework for wildland fire that provides procedural guidelines and incorporates risk-analysis concepts. Conceptually our approach reflects the definition of risk assessment summarised by Fairbrother and Turnley (2005, p. 28): 'Risk assessment uses probabilistic modelling to incorporate environmental stochasticity and experimental uncertainty, and incorporates spatial attributes, simultaneous multiple risks, comparative analyses of different risks, socioeconomic concerns, and ecological effects into the analysis.' The broad structure of the framework was introduced in Fig. 2. A framework for risk-based fire-effects analysis would build on prior similar proposals (Bachmann and Allgower 1998; Shields and Tolhurst 2003) and existing methods, enhanced by systematic advances in fire behaviour and fire-effects sciences. This framework would explicitly incorporate probability analysis at all levels. In the near term there is probably better hope for improving assessment of major first-order effects. The complexity of second-order effects and the uncertainty of future events, e.g. high intensity storms, make prediction of second order inherently more difficult and will likely require long-term research efforts. Although improvements are needed in all stages of the risk-analysis process (Table 1), development is especially needed to improve probability-based analysis of fire effects.

Building a practical framework will require input from many managers and experts and will be most effective if conducted within a common research program similar to a plan recently developed to advance applied wildfire smoke science (A. R. Riebau and D. G. Fox, unpubl. data). Recent roundtables of fire managers and scientists in the United States scoped general needs for a wildfire risk-assessment framework (J. Cissel, pers. comm.) Much more work will be required to: clarify needs, goals, and objectives for risk-based fire-effects analysis; develop analysis process recommendations, methods and guidelines; define science, decision support technology and data requirements; identify existing science, technologies and data that meet framework requirements; and to determine science, technology and data gaps and prioritise improvement needs.

Functional decision-support systems exist to build upon, and meet some general risk-framework requirements. One system provides a set of fuel-treatment planning tools and automatically compiles the spatial data necessary to drive system models for the user-defined area of interest (see Interagency Fuels Treatment Decision Support System, available at http://iftdss.sonomatech.com/, accessed 22 July 2012). A common platform for real-time, spatially explicit wildfire risk analysis and decision support integrates probabilistic fire-behaviour modelling tools and provides web-based mapping services for rapid assessment of values-at-risk (Calkin *et al.* 2011*b*). Both components are supported by nationally consistent databases providing fuels

Table 1. Summary of risk framework components and research needs highlighted in this paper

Fire risk framework components	Research needs
Fire behaviour	Provide fire model outputs relevant to fire-effects prediction
	Improve process-based fire models including geographically extensive validation
	Improve model input data, e.g. fuels and meteorology
Fire effects – 1st order	Process couplings between fire behaviour and fire effects
	Develop fire-effects prediction in herbaceous and shrub vegetation
	Knowledge of soil-surface heat fluxes, physiological basis for tree injury
Fire effects – 2nd order	Improve access to and organisation of fire-effects literature
	Clarify relationship between vegetation change and post-fire erosion
	Changes in sediment flux, biogeochemical cycling and nutrient and constituent export
	Couple forest-regeneration models with models of fire effects
	Develop process-based habitat suitability models with targets linked to fire effects
Spatial and temporal integration	Spatial interactions over time, accounting for system recovery
	Cross-scale analysis over full range of fire-effects interactions
	Assess wildfire disturbance and response relative to other disturbance processes
Risk framework	Methods to quantify expected value change, especially for non-market resources
	Develop probability-assessment techniques accounting for wildfire complexities
	Build seamless, interactive decision-support systems with efficient data management

and weather data to drive fire models and critical infrastructure and occupied-structures data for values assessment. Potential exists for international collaboration and co-development, as similar wildfire decision-support systems are partially or fully implemented and subject to ongoing development in Greece (Kaloudis *et al.* 2005), Australia (Tolhurst *et al.* 2006) and Canada (de Groot 2010; Ohlson *et al.* 2006).

This proposed approach to building a fire-effects analysis framework addresses the limitations of *ad hoc* science, technology and data development and the need for consistent assessment methods. We recommend that a formal working team be assembled to initiate framework development. The information provided in this paper and cited sources may offer a point of departure for their work.

Summary

A comprehensive risk-based approach to fire-effects analysis is needed to support all aspects of wildfire management decisions. However, fire-effects analysis is hindered by significant gaps in core fire behaviour and fire-effects science, limited understanding of the complex spatial and temporal interactions inherent in fire-effects phenomena and lack of a comprehensive plan to address these issues and to build effective decision-support systems. Our survey of science and technology gaps related to prediction of fire effects compliments prior work and reinforces need to develop an integrated wildfire risk-assessment framework. Failure to improve and develop comprehensive methods to predict fire effects will very probably lead to avoidable losses and missed opportunities to restore and improve ecosystems prone to wildfire.

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