CSIRO PUBLISHING

International Journal of Wildland Fire **2014**, *23*, 1045–1060
http://dx.doi.org/10.1071/WF13058

Challenges of assessing fire and burn severity using field measures, remote sensing and modelling

Penelope Morgan[A,E], *Robert E. Keane*[B], *Gregory K. Dillon*[B], *Theresa B. Jain*[C], *Andrew T. Hudak*[C], *Eva C. Karau*[B], *Pamela G. Sikkink*[C], *Zachary A. Holden*[D] *and Eva K. Strand*[A]

[A]University of Idaho, Department of Forest, Rangeland, and Fire Sciences, 875 Perimeter Drive MS 1133, Moscow, ID 83844, USA.
[B]USDA Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory, Missoula, MT 59807, USA.
[C]USDA Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory, Moscow, ID 83843, USA.
[D]USDA Forest Service, Northern Region, Missoula, MT 59807, USA.
[E]Corresponding author. Email: pmorgan@uidaho.edu

Abstract. Comprehensive assessment of ecological change after fires have burned forests and rangelands is important if we are to understand, predict and measure fire effects. We highlight the challenges in effective assessment of fire and burn severity in the field and using both remote sensing and simulation models. We draw on diverse recent research for guidance on assessing fire effects on vegetation and soil using field methods, remote sensing and models. We suggest that instead of collapsing many diverse, complex and interacting fire effects into a single severity index, the effects of fire should be directly measured and then integrated into severity index keys specifically designed for objective severity assessment. Using soil burn severity measures as examples, we highlight best practices for selecting imagery, designing an index, determining timing and deciding what to measure, emphasising continuous variables measureable in the field and from remote sensing. We also urge the development of a severity field assessment database and research to further our understanding of causal mechanisms linking fire and burn severity to conditions before and during fires to support improved models linking fire behaviour and severity and for forecasting effects of future fires.

Additional keywords: fire ecology, fire effects, mapping, remote sensing, retrospective assessment, wildfire environment.

Received 13 April 2013, accepted 14 July 2014, published online 25 November 2014

Introduction

Wildand fires commonly burn extensive areas in forests and rangelands, often jeopardising homes and municipal watersheds, and other places important to society. Fire as a disturbance drives ecosystem composition and structure from sites to landscapes globally (Bowman *et al.* 2009), thus severity is central to evaluating and predicting ecological conditions before, during and after fire. Fire scientists and managers routinely use field and remotely sensed evaluations of fire and burn severity, defined as the magnitude of change due to fire (Lentile *et al.* 2006; Keeley 2009), to describe fire effects on fuels, vegetation, wildlife habitat and soils (Ryan and Noste 1985; Smith 2000; Carey *et al.* 2003; Key and Benson 2006). This information supports strategic planning before and during fires, prioritising post-fire mitigation to diminish flooding and erosion potential and to foster vegetation recovery post-fire (Robichaud *et al.* 2003, 2007*a*; Beschta *et al.* 2004; Kuenzi *et al.* 2008), and making understanding fire and burn severity central to ecologically based fire management.

Despite widespread use, the consistent, objective, repeatable quantification of fire and burn severity remains elusive (Jain *et al.* 2004; Key 2006; Lentile *et al.* 2006; Keeley 2009), and without reliable assessments, the causes and ecological consequences of severity will remain poorly understood. Given the numerous definitions of fire and burn severity, it is important for all users to explain how they define and assess severity (Simard 1991; Lentile *et al.* 2006; Jain *et al.* 2008). As Simard (1991) wrote, 'no two people interpret fire severity the same' because observers focus on fire effects selected for a particular set of local objectives or outcomes. These assessments are rarely impartial, standardised, consistent or comprehensive. Moreover, because fire effects can vary in their scale of impact and temporal recovery, one spatial or temporal scale may not fit all objectives, increasing the challenge of objectively and quantitatively assessing fire or burn severity (Simard 1991; Graham *et al.* 2004). We follow the growing body of literature that differentiates the two severity terms based on temporal and ecological context (Lentile *et al.* 2006; French *et al.* 2008; Veraverbeke *et al.* 2010*a*, 2010*b*). We use 'fire severity' to describe the immediate fire effects and 'burn severity' for the longer-term effects of fires on vegetation and soils (Key and Benson 2006; Lentile *et al.* 2006, 2007; French *et al.* 2008; Veraverbeke *et al.* 2010*a*, 2010*b*). This is consistent with the US National Wildfire Coordinating Group (2012) definition of fire severity as 'degree to which a site has been altered or disrupted by fire, loosely, a product of fire intensity and residence time', but not with the NWCG definition of burn severity as 'soil heating, large fuel and duff consumption, consumption of the litter and organic layer beneath trees and isolated shrubs, and mortality of buried plant parts'; though these clearly influence vegetation response and soil effects (Chafer *et al.* 2004; Chafer 2008; Neary *et al.* 2008). Similar to French *et al.* (2008), we use 'fire and burn severity' or more simply 'severity' unless we refer specifically to immediate fire effects (fire severity) or fire effects measured in the following year or growing season that include some secondary effects and ecological response (burn severity).

Fire and burn severity are often related to fire behaviour either directly or indirectly, but severity is often confused with fire intensity. One major source of misunderstanding about severity occurs when fire behaviour measures are used to characterise severity (Alexander 1982; Ryan and Noste 1985; Keeley 2009). Including fire behaviour measures in severity descriptions is often problematic given that actual fire behaviour measures are likely lacking. Whereas fire behaviour, especially smouldering combustion, is a critical causal mechanism of fire and burn severity, fire behaviour attributes have often been inappropriately used to describe fire effects (Moreno and Oechel 1991). Oliveras *et al.* (2009) found that fire severity was higher in flanking and head fires than backing fires, and Keeley *et al.* (2008) and Jain *et al.* (2004) emphasised that fire behaviour is a critical element in describing fire severity. Thus, fire and burn severity are intimately linked to fire behaviour, but fire behaviour does not fully describe the effect of fire on the ecosystem, especially if it is focussed only on flaming combustion, because it is missing critical ecological responses from the heat of the fire.

Assessing and predicting severity is challenging because fires have multiple effects that are often assessed in different contexts. Assessments of fire and burn severity are done for post-fire mitigation of erosion potential and invasive species establishment, soil erosion potential (Fox *et al.* 2008; Clark and McKinley 2011), post-fire vegetation recovery including tree seedlings (Turner *et al.* 1999; Díaz-Delgado *et al.* 2003; Miller *et al.* 2003; Pausas *et al.* 2003; Beschta *et al.* 2004; Lentile *et al.* 2006), wildlife habitat (Zarriello *et al.* 1995), species of concern (Kotliar *et al.* 2008) and overall vegetation conditions (Bisson *et al.* 2008; Guay 2011). Soil burn severity can also be used to predict the physical, chemical or biological effects (Jain *et al.* 2012), including water repellency (Lewis *et al.* 2006), erodibility (Pierson *et al.* 2001) and nutrient availability (Belillas and Feller 1998). Severity is also a fire regime attribute (Beukema and Kurz 1998; Morgan *et al.* 2001; Barrett *et al.* 2006; Keane *et al.* 2006), perhaps the most difficult one to quantify because in this context it is used more conceptually and lacks measurement units; despite this, no other fire regime attribute is as important in fire ecology. Severity is the basis for a national fire atlas for the US (Eidenshink *et al.* 2007; http://www.mtbs.gov) and has been used to link landscape patterns and scales of disturbance processes (Turner *et al.* 1994; Chuvieco 1999; Hudak *et al.* 2007*a*, 2007*b*). Managers and scientists use fire and burn severity classifications to evaluate prescribed fire success (Ryan and Noste 1985), stratify post-fire vegetation and soil response, and describe burn patterns (Carey *et al.* 2003).

Despite the extensive scientific literature describing fire effects and elements of severity in different ecosystems, there are few widely accepted or standardised measures of severity consistently applicable within such different assessment contexts (Halofsky and Hibbs 2009). Severity is variously used as a concept, a continuous variable, a class and an index. Ideally, metrics of severity should be specific, meaningful and readily interpretable, as well as measureable in the field and remotely, and at multiple spatial and temporal scales (Hudak *et al.* 2007*b*). Satellite imagery, statistical and simulation modelling, and standardised efficient field sampling have facilitated the generation of quick and inexpensive fire and burn severity maps, minimising the need for extensive resource-intensive, and potentially dangerous, post-fire field sampling. However, all of these advances are limited by ecological, technical and logistical issues.

We address four objectives in this paper. We examine recent advances in predicting and assessing fire and burn severity in the field, remotely and using models. We discuss the numerous factors and associated interactions that influence severity and challenge our ability to consistently assess or predict it. We provide guidance for those who wish to use severity assessments in planning and implementing land management activities, using vegetation and soil burn severity as examples. We propose a strategy for describing fire and burn severity in the future to reduce the confusion and complexity of defining, measuring and assessing severity, while providing the ability to design severity assessments for specific uses. Our focus is on new research building upon previous work conducted by Lentile *et al.* (2006) and Keeley (2009).

Assessing severity

Assessing fire and burn severity in the field

Many fire effects have been measured to describe fire and burn severity (French *et al.* 2008; Jain *et al.* 2012) (Table 1). Changes in overstorey trees or shrubs are the primary fire effect implicit in most fire and burn severity assessments (Regelbrugge and Smith 1994; Patterson and Yool 1998; Turner *et al.* 1999; Smith *et al.* 2007). However, assessing plant mortality can be somewhat problematic in ecosystems where the dominant plants often re-sprout quickly after fire, such as in many hardwood forests, shrublands and grasslands. Therefore, Keeley (2009) advocated using the amount of biomass consumed instead of plant mortality, and Wang and Glenn (2009) used changes in height to measure severity in shrublands. Surface fuel consumption has also been used as an important indicator of severity (Schimmel and Granstrom 1996; Boby *et al.* 2010;

Table 1. Quantifying burn severity for forest soils (Jain *et al.* 2012)

Each subject area includes the literature sources, application, number of categories identified and range of possible post-fire outcomes. Cells with dashes (–) denote an outcome that was not included. Post-fire characteristics most noted had two primary indicators: first, the amount of pre-fire surface organic matter (e.g., litter, humus, rotten wood) present, expressed as abundant, present or absent on the forest floor; and second, whether the litter was scorched (S) and the assessed state of the exposed mineral soil – unburned (U), blackened from combustion (B), gray/white (G) from ash or orange (O) from mineralogical changes and fire residuals. Numbers in the table indicate the level of severity used by that classification, where 0 is unburned and included as a class, and 1–5 represents relative low to high severity.

Source	Application	Number of categories	Pre-fire	Post-fire characteristic Forest Floor Surface Organic Matter Condition (S – scorched, U – unburned, B – black, G – grey/white, O – orange)												
			Unburned	Abundant					Present				Absent			
				S	U	B	G	O	U	B	G	O	U	B	G	O
Physical effects																
Johansen *et al.* (2001)	Erosion	2	–	1	1	1	1	1	1	1	1	1	2	2	2	2
Neff *et al.* (2005)	Erosion	2	0	2	2	2	2	2	2	2	2	2	2	2	2	2
Parsons *et al.* (2010)	Values at risk	3	–	1	1	1	1	1	3	3	3	3	5	5	5	5
Robichaud and Hungerford (2000)	Water infiltration	2	–	1	1	1	1	1	2	2	2	2	2	2	2	2
Shakesby *et al.* (2003)	Water repellency	2	–	1	1	1	1	1	1	1	1	1	2	2	2	2
Ulery and Graham (1993)	Physical	4	–	1	–	–	–	–	–	–	–	–	1	2	4	5
Chemical effects																
Arocena and Opio (2003)	Mineralogy	2	0	–	–	–	–	–	–	–	–	–	–	–	2	–
Baird *et al.* (1999)	Nitrogen	2	0	2	2	2	2	2	2	2	2	2	2	2	2	2
Brais *et al.* (2000)	Chemistry	2	–	1	1	1	1	1	1	1	1	1	2	2	2	2
Cerri *et al.* (1991)	Nutrient dynamics	2	0	–	–	–	–	–	–	–	–	–	–	2	2	2
Ellingson *et al.* (2000)	Nitrogen	2	0	–	–	–	–	–	–	–	–	–	2	2	2	2
Rumpel *et al.* (2007)	Chemistry	3	0	–	–	–	–	–	3	3	3	3	5	5	5	5
Yeager *et al.* (2005)	Nitrogen fixation	3	0	–	–	–	–	–	–	–	3	3	–	–	5	–
Biological effects																
Bentley and Fenner (1958)	Seed survival	5	–	1	1	1	1	1	2	2	2	2	3	4	5	–
Bernhardt *et al.* (2011)	Vegetation	3	–	1	1	1	1	1	3	3	3	3	5	5	5	5
Blank *et al.* (1994)	Vegetation	2	0	2	2	2	2	2	2	2	2	2	2	2	2	2
Bonnet *et al.* (2005)	Vegetation	4	0	3	4	5	5	5	4	5	5	5	5	5	5	5
Choromanska and DeLuca (2002)	Microbes	2	0	–	–	–	–	–	–	–	–	–	2	2	2	2
Dyrness and Norum (1983)	Biological review	5	0	2	3	3	3	3	4	4	4	4	–	–	5	–
Jain *et al.* 2006; Jain and Graham (2007)	Forest structure	4	0	3	3	3	3	3	4	4	4	4	5	5	5	5
Larrivée *et al.* (2005)	Arthropods	2	0	2	2	2	2	2	2	2	2	2	–	–	–	–
Lentile *et al.* (2005)	Regeneration	3	–	1	3	3	3	3	3	3	3	3	5	5	5	5
Morgan and Neuenschwander (1988)	Shrubs	2	–	1	1	1	1	1	–	–	–	–	2	2	2	2
Schimmel and Granstrom (1996)	Seed survival	3	0	1	1	1	1	1	3	3	3	3	5	5	5	5
Tyler (1995)	Vegetation	2	0	1	1	1	1	1	2	2	2	2	2	2	2	2
Wang and Kemball (2005)	Seed survival	3	–	1	2	2	2	2	–	–	–	–	5	5	5	5
Assessments																
Alexander *et al.* (2006)	Physical setting	3	–	1	1	1	1	1	3	3	3	3	3	5	5	5
Barkley (2006)	Monitoring	3	–	1	1	1	1	1	3	3	3	3	5	5	5	5
Chafer *et al.* (2004)	Remote sensing	5	–	1	1	2	2	2	3	3	3	3	4	5	5	5
Key and Benson (2006)	Remote sensing	4	0	2	2	2	2	2	3	3	3	3	5	5	5	5
Lutes *et al.* (2006)	Monitoring	5	0	1	2	2	2	2	2	3	3	3	3	5	5	5
Miller (2001)	Monitoring	5	0	2	2	2	2	3	4	4	3	4	4	5	5	5
Patterson and Yool (1998)	Remote sensing	4	0	2	2	2	2	2	4	4	4	4	4	5	5	5
Ryan and Noste (1985)	Prescribed fire	4	0	2	2	2	2	3	3	4	4	4	4	5	5	5
US Department of Interior (2003) (forests, shrublands)	Monitoring	5	0	2	3	3	3	3	4	4	4	4	4	5	5	5
US Department of Interior (2003) (grasslands)	Monitoring	5	0	2	3	3	3	3	–	–	–	4	4	5	5	5
White *et al.* 1996	Remote sensing	3	–	1	1	1	1	1	1	1	1	1	3	5	5	5

Table 2. Variables to measure in assessing burn severity

Those variables in bold are most commonly measured and can be more readily inferred from satellite imagery. To measure fire severity for a particular objective, select those fire effects important for assessing the objective. Use continuous variables whenever practical: these can always be collapsed into classes if needed; for example, for interpretation. Name the resulting fire severity index for the objective, such as soil severity index (see examples in Table 1 and Table 3). From Jain *et al.* (2012)

Selected fire effect	Measurement variable	Calculation	Description
Plant mortality	Percentage dead (%)	Trees >5 cm DBH	Fire-caused overstorey tree mortality
		Trees <5 cm DBH	Fire-caused understorey tree mortality
	Reduction in cover (%)	Shrubs	Shrub cover reduction
		Herbs	Herbaceous cover reduction
Fuel consumption	Reduction in loading (%)	Woody	Proportion woody fuel consumed
		Duff	Proportion duff fuel consumed
	Consumption (kg m^{-2})	Woody	Amount of woody fuel consumed
		Duff	Amount of duff fuel consumed
Smoke	Emissions (kg km^{-2})	PM2.5	Amount of particulate matter released
		CO_2	Amount of carbon dioxide emitted
Char fraction	Char fraction	Percentage	Amount of char in soil
Soil heating[1]	Depth lethal heating (cm)	Depth >60°C for 1 min	Soil depth of tissue death
	Depth nutrient heating (cm)	Depth >250°C	Soil depth of nutrient changes
	Total heat	Integrated area >60°C under time–temperature curve at 2 cm	Total soil heating
Soil water Repellency[1]	Soil infiltration rate (mm hr^{-1})	Rate (mm hr^{-1}) measured with mini disk infiltrometer	Soil infiltration conditions
Nutrients[1]	Reduction in nitrogen (%)	NH_4 concentration before and after fire	Difference between pre- and post-fire NH_4 concentration
Erosion[1]	Exposed mineral soil	Percentage based on visual estimate	Amount of ground area in mineral soil

[1]Not readily inferred from remotely sensed imagery.

Keane *et al.* 2010; Hudak *et al.* 2013), and changes in soil properties, such as water repellency (Lewis *et al.* 2006), erodibility (Pierson *et al.* 2001) and nutrient availability (Belillas and Feller 1998) have been the focus in other studies.

The composite burn index (CBI) is one field measurement that has been widely used in burn severity assessments, especially for vegetation (Key and Benson 2006; Miller and Thode 2007; Holden *et al.* 2009; Miller *et al.* 2009; Soverel *et al.* 2010; Dillon *et al.* 2011; Cansler and McKenzie 2012). CBI was designed to correlate rapid field assessments of fire effects on vegetation and soils with the difference between pre- and post-fire satellite images (Key and Benson 2006). However, CBI is an integrated metric that averages the magnitude of change across five strata from soil to vegetation canopies, with each strata having four or five variables that are visually assessed and assigned a value between zero (unburned) and three (highest severity). As such the specific factors resulting in a given CBI value can become obscured. CBI is heavily weighted to measuring fire effects on vegetation (Miller and Thode 2007) and correlates most closely with changes in the upper canopy structure in forests (Miller and Thode 2007; Miller *et al.* 2009). To improve the correlation between ground measurements and remotely sensed data, De Santis and Chuvieco (2009) developed the GeoCBI, which adjusts the weighting of each stratum according to its fractional cover as viewed from overhead. Although the CBI and GeoCBI have been applied successfully in a wide variety of ecosystems, CBI performs poorly in ecosystems like Alaskan boreal forests (French *et al.* 2008) and California chaparral (Keeley *et al.* 2008), where severity is best represented respectively by depth of burn in soil organic matter or amount of shrub biomass consumed. CBI and GeoCBI can be correlated with spectral reflectance in a satellite image, but it can be difficult to interpret the specific fire effects that are present using these integrated indices. Further, while visual estimates are the only practical method of estimating most fire effects that comprise the CBI, these estimates have subjective bias across users and site conditions that make consistent and accurate evaluation difficult. Potentially great differences in perceptions and interpretations among observers with varying levels of experience can confound consistent CBI visual evaluations of severity attributes across different ecosystems. Plant species, soil types and fuelbeds are also different across large fires making consistent evaluations problematic.

Burn severity is commonly assessed in the field and then combined with remote sensing (Table 2). Parsons *et al.* (2010) developed a quantitative method for assessing soil burn severity. They focussed on soil effects to assist interpretation of Burned Area Reflectance Classification (BARC) maps (Clark and McKinley 2011; http://www.fs.fed.us/eng/rsac/baer/barc.html) in post-fire assessment and rehabilitation. They used five factors to help validate the BARC maps, which include ground cover remaining, ash colour and depth, fine roots remaining, soil structure changes, and water repellency. Like the soil post-fire index developed by Jain *et al.* (2012) (Table 3), the assessments are made in the field with continuous measures of importance to fire effects on soils. A developing database (P.R. Robichaud, pers. comm.) of these continuous measures assessed in the field immediately after fire will be immensely helpful in future evaluations of fire and burn severity from satellite imagery.

Table 3. Soil post-fire index (PFI) classification key developed by Jain *et al.* (2008) based on (1) the abundance of surface organic matter to create broad categories and (2) mineral soil colour to partition the broad categories

Surface organic cover can include litter, humus, rotten wood and in some cases a root mat. As with any key, one begins by evaluating the site based on 1a or 1b. If the response to 1b is 'yes' then surface organic cover is evaluated using ocular or grid sampling estimates. For example, if surface organic cover is <40% (3b) than this broad category can be dissected based on mineral soil colour (5a–5d). If the plurality of the soil is charred orange (5d) then the resulting soil PFI is 3.4

Soil characteristics		Soil PFI category
1a	No evidence of a recent fire	0.0
1b	Evidence of recent fire	
2a	Surface organic cover ≥85%	1.0
2b	Surface organic cover <85%	
3a	Forest floor surface organic cover ≥40% and mineral soil appearance has a plurality of:	2.0
4a	Unburned mineral soil	2.1
4b	Black charred mineral soil	2.2
4c	Grey or white charred mineral soil	2.3
4d	Orange charred mineral soil	2.4
3b	Surface organic cover <40% and mineral soil appearance has a plurality of:	3.0
5a	Unburned mineral soil	3.1
5b	Black charred mineral soil	3.2
5c	Grey or white charred mineral soil	3.3
5d	Orange charred mineral soil	3.4
3c	(Forest floor absent) No surface organic matter left and mineral soil appearance has a plurality of:	4.0
6a	Unburned mineral soil	4.1
6b	Black charred mineral soil	4.2
6c	Grey or white charred mineral soil	4.3
6d	Orange charred mineral soil	4.4

Perhaps the greatest challenge to severity evaluations in the field is that without pre-fire measurements, changes due to fire must be inferred retrospectively from post-fire conditions; this is the case in most assessments of severity following wildfires (Hudak *et al.* 2011, 2013). In temperate forests, common fire effects measured to evaluate severity include the amount of surface fuel consumed; percentage mortality in both overstorey trees and understorey plants; percentage tree or shrub volume and cover affected and inferred degree of soil heating. In grasslands and boreal forests, the degree of soil heating and depth of burning, including percentage consumption of soil organic matter, are important fire effects (French *et al.* 2008). However, for most of these effects, both pre- and post-burn measurements are required to accurately and objectively quantify the change caused by fire. Unfortunately, given that wildfire locations are difficult to predict, pre-burn measurements are typically lacking and are at best challenging to acquire or infer (Lentile *et al.* 2007).

Several strategies are used to address the lack of pre-fire measurements. Most often, observers subjectively estimate pre-fire conditions (e.g. canopy cover of understorey and overstorey plants, duff and litter cover, and surface fuel load) based on observations of surrounding areas (Key and Benson 2006). Burned plots can be compared with paired plots in adjacent unburned areas (Díaz-Delgado *et al.* 2003; Karau and Keane 2010; Hudak *et al.* 2011), but these inferences depend on how well the unburned plot represents the pre-fire condition of the burned plot. Sometimes, sites have been sampled pre-burn, such as the network of Forest Inventory and Analysis (FIA) plots across all forested lands in the US (Megown *et al.* 2011), but rarely are the number of burned FIA plots for a given wildfire sufficient for a statistically valid fire effects evaluation.

In some instances, severity can be inferred from post-fire evidence alone. Based on post-fire diameter measurements of all live and dead trees in conifer forests, Miller *et al.* (2009) were able to calculate pre- and post-fire live tree basal area, and subsequently a measure of basal area loss due to fire. In chaparral vegetation, the diameter of the smallest branch remaining on shrub skeletons has been found to be a good indicator of overall biomass loss from fire (Moreno and Oechel 1989; Keeley *et al.* 2008). In other forest studies, the amount of charred surface fuels, soil and trees is sometimes used to evaluate respectively the magnitude of fuel consumption, soil heating and plant mortality. However, this evidence does not provide a complete picture or a true quantitative assessment of the ecological consequences of the fire. Tree bole char, for example, is only partially correlated with tree mortality (Keyser *et al.* 2006; Halofsky and Hibbs 2009; Hudak *et al.* 2011). In rangelands, high fire severity has been associated with reduction of the seedbank, lower species diversity post-fire and increases in exotic species cover (Ghermandi *et al.* 2013).

Assessing fire and burn severity with remote sensing

Remotely sensed image data have the great advantage of providing pre-fire information that can be difficult or impossible to retrieve in the field. Surface reflectance changes over the days and weeks following fires (Trigg and Flasse 2000), and fires themselves, change surface reflectance in a wide variety of ecosystems (Landmann 2003, Chafer *et al.* 2004; Cocke *et al.* 2005; Smith *et al.* 2005; Roy *et al.* 2006; Chafer 2008; French *et al.* 2008; Kumar *et al.* 2008; Lee *et al.* 2009; Veraverbeke *et al.* 2011). Many severity assessments use satellite imagery to quantify the magnitude of vegetation change from pre-fire conditions (e.g. Key and Benson 2006; Miller and Thode 2007; Holden *et al.* 2009; Miller *et al.* 2009; Soverel *et al.* 2010). If the same type of sensor is used to collect imagery both before and after the fire under comparable illumination conditions, then the difference between them can provide an objective means to quantify ecological change induced by the fire, which has a substantial effect on the reflective properties of the scene (Jakubauskas *et al.* 1990; Landmann 2003). This is the basis for using the Normalized Burn Ratio (NBR, Key and Benson 2006), Normalized Difference Vegetation Index (NDVI, Tucker 1979) and similar indices. Many calculate differences between pre- and post-fire indices in absolute (differenced NBR or dNBR) or relative terms as relative differenced Normalized Burn Ratio (RdNBR, Miller and Thode 2007), the Relativized Burn Ratio (RBR), or differenced NDVI (dNDVI, Díaz-Delgado *et al.* 2003; Epting *et al.* 2005), particularly where the pre-burn biomass is low or highly variable. Although the dNBR is used more broadly as a burn severity index, it sometimes performs only marginally better than dNDVI (Hudak *et al.* 2007*b*; Veraverbeke *et al.* 2010*b*). Fox *et al.* (2008) found NBR and NDVI are highly correlated. Roy *et al.* (2006) advised caution in relying on NBR and related indices because the resulting burn severity maps may have low accuracy depending on the variable of interest and whether the index was originally developed for detecting burned area, not burn severity. Clearly, each assessment of burn severity must be carefully evaluated to ensure it meets the intended goal with acceptable accuracy.

The Landsat sensors have provided the longest available and most widely interpreted source of image data for assessing severity using the indices detailed above. Fire and burn severity maps derived from Landsat Thematic Mapper (TM) imagery can be used to develop retrospective maps of historical wildland fires back to 1984 (Eidenshink *et al.* 2007). Landsat Multi-Spectral Scanner (MSS) imagery (Russell-Smith *et al.* 1997; Hudak and Brockett 2004) and aerial photographs (Ekstrand 1994; White *et al.* 1996) can extend these records even further back in time. Severity mapping from imagery usually involves relating spectral reflectance characteristics of the post-fire scene to field measures of fire effects or severity indices (e.g. CBI) collected at coincident locations (Cocke *et al.* 2005; Hudak *et al.* 2007*b*). Strong correlations between the datasets can then be interpreted to indicate the field variable of interest. While 30-m resolution Landsat TM imagery is most commonly used to map fire-induced change, higher spatial and spectral resolution data have obvious potential for quantifying fine-scale post-fire effects.

Other types of imagery are used to quantify severity. Robichaud *et al.* (2007*b*) used hyperspectral imagery of higher spatial and spectral resolution than Landsat TM imagery to more accurately map post-fire soil and ash cover fractions for assessment of erosion potential. Moreover, they associated the high resolution imagery to directly measured variables instead of indices. However, high resolution multispectral and hyperspectral data are expensive and can be challenging to work with because they contain hundreds of spectral bands, have relatively high densities of pixels and may require extensive image geo-registration. These factors may limit operational use of hyperspectral data for mapping post-fire effects. MODIS imagery, which is freely available like Landsat data but of much coarser resolution (250–1000 m), can be used to assess severity of large wildfires when other data are unavailable (Kolden and Rogan 2013), but the coarse spatial resolution makes the resulting maps unsatisfying to most managers and scientists.

Active sensor systems, such as Light Detection and Ranging (LiDAR) and radar can also be used to infer fire effects. In contrast to passive sensors measuring the sun's reflected radiation these sensors supply their own power. LiDAR data have demonstrated potential for characterising fire-induced changes in overstorey vegetation characteristics (Wang and Glenn 2009; Wulder *et al.* 2009; Kwak *et al.* 2010; Magnussen and Wulder 2012), For example, LiDAR has been used for post-fire assessments to quantify tree regeneration (Debouk *et al.* 2013), assess how post-fire forest structure varies with burn severity (Kane *et al.* 2013), and estimate post-fire tree height (Magnussen and Wulder 2012). However, operational use of LiDAR remains limited given the sparse coverage of pre-fire data and the expense of acquiring new post-fire data. Radar data may also have potential for severity assessments, but difficulty in data interpretation limits operational utility (Kasischke *et al.* 2007*b*; Tanase *et al.* 2010*a*; Tanase *et al.* 2010*b*; Tanase *et al.* 2010*c*).

Ultimately, to maximise the utility and efficiency of remote sensing assessments, the spatial, temporal and spectral resolution of the remotely sensed imagery must match the ecological scale of the fire effect of interest. More research into active sensors would likely lead to more physically based severity assessments, which are needed to advance fire science. However, approaches that directly estimate biophysical measures of interest can also be applied to passive optical imagery, which would reduce current overreliance on burn severity indices to infer biophysical attributes (Roy and Landmann 2005; Kasischke *et al.* 2007*a*). Spectral mixture analysis is an appealing approach to processing post-fire satellite imagery because it estimates the fractional cover of biophysical variables at the sub-pixel level, making them more directly comparable to the same fractional cover variables measured in field plots on the ground. For instance, estimates of char fraction or green vegetation fraction derived from imagery correlate as well as NBR to the same fractional cover variables estimated independently on the ground (Hudak *et al.* 2007*b*; Smith *et al.* 2007). Others have estimated severity using approaches that incorporate radiative transfer modelling, which takes advantage of specific physical reflectance properties of surfaces to estimate vegetation parameters (Chuvieco *et al.* 2006, 2007). As applied to burn severity mapping, these efforts use a reference spectrum of a range of healthy to damaged vegetation, then invert a radiative transfer model to simulate post-fire spectral response, and finally apply a supervised classification to a post-fire image (De Santis *et al.* 2009; De Santis *et al.* 2010). This method has the capacity to establish scenarios for combinations of vertical strata severities that are less dependent on particular local conditions. It can also simulate the outcome of specific effects of fire, such as changes in leaf colour or leaf area index because they are input variables in the canopy reflectance models (De Santis *et al.* 2009).

Predicting fire and burn severity with modelling

Statistical and simulation modelling are alternatives to map and assess post-wildfire severity. Statistical relationships to predict severity can provide insight into possible drivers of severity.

Holden *et al.* (2009) created a statistical severity model from field-gathered CBI data, landscape and biophysical spatial data, and NBR from Landsat imagery in New Mexico, USA. Dillon *et al.* (2011) expanded on this statistical approach by adding weather and hydroclimatic indices to predict proportion burned with high severity and to map areas that could burn severely if a wildfire were to occur for the western US. In South Korea, Lee *et al.* (2009) used regression tree analysis to predict severity from landscape characteristics. Simulation models such as FOFEM (Reinhardt *et al.* 1997), and statistical models such as CONSUME (Ottmar *et al.* 1993) are useful for simulating the direct effects of a fire on vegetation, fuels and soils. Keane *et al.* (2010) implemented FOFEM into a spatial computer application called FIREHARM to create severity maps, which have been compared with and integrated with satellite imagery (Karau and Keane 2010; Karau *et al.* 2014). These types of models have several advantages: (1) simulation models can provide biophysically based fire effects estimates, (2) results can be scaled to the resolution most appropriate for describing a specific effect, (3) models allow for rapid assessment because results can be simulated quickly as long as input data are available, and (4) models can be used to predict fire effects, allowing a manager to proactively prioritise resources.

The disadvantage of current empirical and simulation models is that severity predictions are only as good as the input data used to create them (Karau *et al.* 2014), and the widely available spatial data used to develop the simulation and statistical models have high levels of uncertainty (Keane *et al.* 2013). Current models rarely use data from ongoing severity assessments. Most severity modelling efforts use a completely different set of severity assessments and classifications than remote sensing or field methods. Therefore, a standardised set of ecological metrics to calibrate mapped severity to locally observed fire effects is critical for all three severity methods – empirical, imagery and simulation.

Challenges common to all assessments of fire and burn severity

Spatial variability

Fires do not burn homogeneously across landscapes (Fig. 1), nor do they burn similarly in different ecosystems, giving rise to challenges with characterising fire and burn severity. Understanding the causes of this heterogeneity is often a goal of severity assessments. In this context, severity has been linked to many biophysical characteristics and processes (Keeley 2009). Landscape patterns of severity often vary with topography (e.g. Kushla and Ripple 1997; Broncano and Retana 2004; Holden *et al.* 2009) because topography influences the biophysical environment (microclimatic conditions of temperature, precipitation, direct solar radiation, timing of snowmelt, wind exposure) that directly affects both biomass accumulation and the amount of biomass available to burn at the time of fire (Holden and Jolly 2011). Dillon *et al.* (2011) found the distribution of high-severity effects on forests in six regions in western US fires was influenced more strongly by topographic

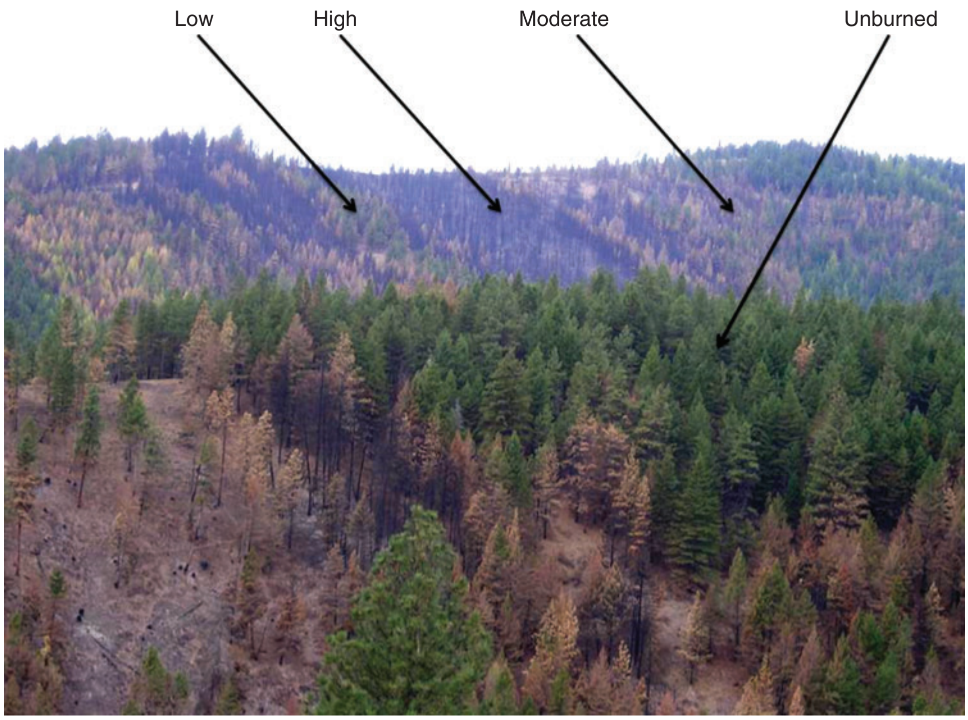

Fig. 1. Post-fire landscape near the lightning ignition start of the 2003 Black Mountain 2 fire near Missoula, Montana, 8 weeks after burning. The heterogeneous pattern of fire severity patches is a result of the interactions of fire with the biotic and physical environment at multiple scales. As time since fire progresses, the relative differences in fire severity, and their spatial pattern, will change.

characteristics than weather or climate, but the specific effects of topography varied geographically. Similarly, Broncano and Retana (2004) found spatial variability in severity to be correlated with elevation and aspect in Spain, and Chafer *et al.* (2004) and Bradstock *et al.* (2010) found that topographic setting, combined with fuel loading and fire weather, influenced severity patterns in Australia.

Unlike topography, which is relatively static and easily quantified pre-fire, quantitative measures of pre-fire fuel and local weather conditions on the ground during a fire are rarely available and difficult to infer from post-fire assessments. Pre-fire character of the vegetation community, such as structure (density, size, height), composition (fire-adapted species *v.* fire-sensitive species), and productivity (biomass, deposition), moisture content and phenology, sets the stage for a range of possible fire behaviours and ecological responses. Severity might be considered low in a grassland but high in a forest; thus the magnitude of severity will vary depending on what is burned. However, there are studies that show a relationship between pre-fire structure and severity. Jain *et al.* (2006) and Jain and Graham (2004) demonstrated close relationships between soil burn severity and both pre- and post-burn forest structure. Hessburg *et al.* (2007) related severity to forest structure across landscapes in the north-western US. Bigler *et al.* (2005) found that severity correlated with forest structure and composition, stand history (previous disturbances) and elevation.

Previous disturbance histories, climates and antecedent weather create vegetation conditions that influence the complex web of fire effects that can be spatially heterogeneous, making it difficult to predict severity (Turner *et al.* 1999; Romme 2005; Thompson and Spies 2010). As a fire burns, the micrometeorology of fire weather interacts with ignition and combustion to leave behind unique patterns of fire effects. Subtle changes between day- and night-time temperatures and humidity can alter fire behaviour and severity. Wind during a fire event can influence severity patterns even in relatively uniform vegetation. The presence or absence of ladder fuels may also determine the severity outcome and forest composition and structures can be influenced by harvest, disease and insect infestations. Tree mortality, for example, is a result of a complex set of interactions between instantaneous fire behaviour, topography, ambient weather conditions and tree characteristics including size, crown position, adjacency of neighbouring trees, bark thickness and other factors influencing the ability of the tree to survive the fire – all of which vary at different scales. These interacting processes and scales make severity estimation difficult because the elements that dictate fire and burn severity may vary at scales that differ by ecosystem, fire and biophysical environment.

Ecological responses occur at multiple time scales and this can confound fire and burn severity assessments (Lentile *et al.* 2006). Pre- and post-fire precipitation patterns, wind events, human interventions, and plant reproductive strategies all influence long-term ecosystem response to fire yet they act at different temporal scales, which adds uncertainty to assessments of fire severity. Changes in post-fire soil nitrogen, for example, last for a shorter time (years) than changes in forest structure (decades). Soil water repellency usually lasts only a few years (Doerr *et al.* 2000; Robichaud and Hungerford 2000), but recovery of vegetation can take decades.

Multiple interacting fire effects

The major factors used to assess severity, such as plant mortality and soil heating, are not independent ecological processes but are interrelated through mutual feedback mechanisms that vary greatly between fuel and vegetation types. Grass fires can have high intensities, rapid rates of spread and nearly complete fuel consumption, yet soil heating and plant mortality are usually low and vegetation conditions 1-year post-fire can be similar to that found before the fire. In contrast, a low-intensity surface fire burning in a stand of fire-intolerant spruce and fir trees can kill many trees through cambial damage on stems and roots through soil heating. Tree mortality, for example, often increases when insects and disease agents attack trees weakened by fire (Hood *et al.* 2007), sometimes enough to alter post-fire assessments of severity. Conversely, soil erosion potential after fire may be mitigated by needles from scorched trees that fall on severely burned, highly erodible soils (Pannkuk *et al.* 2000). High consumption of aboveground biomass in perennial grasslands and shrublands may be short lived with sprouting species recovering relatively quickly post-fire. Fires that increase non-native plant species cover may be more severe than fires where only native plants are present. In some instances, it may be necessary, and even desirable, to account for some secondary fire effects in fire and burn severity assessments (Veraverbeke *et al.* 2010*a*; 2010*b*, 2011; Dillon *et al.* 2011).

Spatial and temporal scale

Scale considerations are essential for appropriate assessment of fire effects (Simard 1991). Wildland fire acts across multiple temporal and spatial scales responding to factors that control both fire behaviour (e.g. fuel moisture, wind) and the characteristics of the biological elements that are burned (e.g. species, size, loadings) (King *et al.* 2008). These interactions in turn influence severity. As the factors that control fire and vegetation act across different scales, it follows that fire and burn severity must be described across multiple time and space scales. Further, spatial pattern influences vegetation response – if patches of high severity are very large, recovery of vegetation dependent on dispersal of seeds from surviving plants will be slower than in a fine-scale mosaic (Turner *et al.* 1999; Bonnet *et al.* 2005; Donato *et al.* 2009). Thus, metrics of the spatial distribution of fire effects are needed to fully quantify severity.

The timing of specific fire effects and ecosystem responses can be dramatically different within a single fire. Vegetation recovery, for example, can take only a few years in low-elevation grasslands, but may take decades in upper subalpine forests (Keane and Parsons 2010). Similarly, tree regeneration may occur quickly after fire in productive mesic forests, but may be slower in xeric, cold upper subalpine environments (Agee and Smith 1984). This difference in response timing serves to complicate many burn severity assessments.

Recommended best practices

Instead of collapsing complex fire–biota–environmental interactions and responses down to a generalised classification, we recommend directly measuring the actual fire effect, be it tree mortality, fuel consumption, soil water repellency or any other important measureable fire effect. As Jain *et al.* (2004) emphasised, researchers should simply quantify severity with what they are actually measuring (e.g. see Table 3). The soil burn severity index is a good example that is widely applied in assessing post-fire effects on soils with a focus on soil erosion potential. Another is the soil PFI based on post-fire characteristics that relate to nutrient availability, seed availability and other soil characteristics (Jain *et al.* 2012) (Table 3). These are good examples that use physically based fire effects variables of interest that can then be input into other fire effects applications (erosion modelling, wildlife habitat evaluations) for a more tailored assessment of severity, and they can be predicted from simulation models to expand the use of the severity index from operational to planning and from only retrospective to predictive. A common database of severity assessments will be immensely helpful for improving inferences beyond local applications. Further, we urge use of continuous variables for measurement whenever practical; these can always be collapsed to classes if needed for interpretation.

Every assessment will require addressing questions of which imagery, which indices, what timing and what to measure, as illustrated in Table 4. We outline best practices as these choices are made for assessing soil burn severity and vegetation effects. Imagery choice is often dependent on what is available. For burn severity assessments, Landsat TM and ETM+ sensors are often used because of the 30-m spatial resolution, and the global availability of the imagery every 16 days and large catalogue of free images dating back to 1984, all of which are important for rapid post-fire assessments needed for mitigating erosion potential. Clearly, soil effects vary at scales finer than 30 m (Hudak *et al.* 2007*b*) and degree of soil charcoal and organic content of soils can complicate satellite-inferred burn severity (Epting *et al.* 2005; Smith *et al.* 2010; Picotte and Robertson 2011). For vegetation effects, Landsat imagery is commonly used, but other imagery products with finer spatial and spectral resolution is available. NBR, dNBR and RdNBR are most common, but RBR and char fraction (Lentile *et al.* 2009) may be better suited depending on the specific fire effects of interest. Relativised measures, including both RdNBR (Miller and Thode 2007) and RBR are better for detecting high-severity effects across a wide range of pre-fire conditions, including those with low total biomass.

Timing of imagery depends on the purpose of assessment. When choosing pre- and post-fire images, it is important to consider that vegetation that burns often does so in a drought-stressed state. Therefore, the pre-fire image must be collected as close to the fire date as possible to isolate the effects of fire from the effects of drought. Timing of the post-fire image depends on several considerations, including ecological context and the specific purpose of the assessment. For instance, how quickly the vegetation will respond or recover from the fire must be considered. Is it important to capture immediate, same-season effects before any recovery or is it desirable to allow time for some second-order effects and initial recovery? Has the vegetation senesced or did snow fall immediately post-fire, making change detection impossible? If next growing season imagery is required, when will phenology most closely match the pre-fire image? In rangelands, remotely sensed reflectance is highly variable with phenology throughout the

Table 4. Guidance for assessing burn severity for vegetation effects and soil burn severity using field and remote sensing methods

For each application, users need to decide which imagery and index, the timing, and what to assess in the field: see text, Eidenshink *et al.* (2007) for further discussion and references for further information. The resulting severity indices would differ from each other and from those developed for assessing fire effects on habitat for invasive species, wildlife or other purposes

Question	Soil burn severity	Vegetation severity
What imagery?	• LANDSAT most commonly used due to availability, spatial resolution and cost. • Quickbird or other high spatial resolution imagery useful when higher resolution is needed, but costs more. MODIS over larger extents where lower spatial resolution is acceptable.	
What index?	• NBR (one image immediately post-fire), dNBR or RdNBR or RBR • Adjust based on field assessments.	• Relativised measures (e.g. RdNBR or RBR) commonly used, especially for areas with relatively low or heterogeneous vegetation cover; dNDVI useful
Timing of imagery?	• Immediately post-fire to support planning for rehabilitation and recovery	• Usually extended with pre-fire image as close to fire date as possible, post-fire image 1 year post-fire at same phenology, but with rapid vegetation recovery use imagery immediately post-fire • For non-forests, often immediately pre- and immediately post-fire
Field measures	• Focus on direct measures such as soil colour and exposure, and water repellency. Indirect measures include fuel consumption and amount of ash	• Depends on purpose of assessment (see Table 2) • If field measures will be used in combination with remote sensing, then only measure variables that can be readily inferred from imagery and match the spatial and temporal scale • CBI or GeoCBI commonly used but we recommend measuring the actual effect(s) of interest directly using quantitative, continuous measures where possible. Examples include tree mortality, fuel consumption, proportion of foliar biomass burned and reduction in canopy cover

growing season, which must be considered when selecting images for burn severity assessments.

It is important to think carefully about field measures, especially if these are to be inferred from satellite imagery or linked to predictive models. If field measurements are to be correlated with remote sensing data, the variables measured in the field must have a logical and mechanistic connection to properties the sensor can detect. For instance, soil heating by fire, although ecologically important, cannot be inferred directly from pre- and post-fire satellite imagery comparisons. Correlations between field and remotely sensed variables say nothing about causation; remotely sensed indices of severity are only indices, and therefore should not be interpreted as direct measures of fire or burn severity.

Future directions

We suggest that the first step for improving severity assessments is to move towards a unified, physically based, hierarchical terminology (Table 2). Fire and burn severity are general concepts to qualitatively or quantitatively describe the magnitude of the myriad immediate and longer-term fire effects at a point, plot, stand and across a landscape (Fig. 2). Remote sensing indices (Fig. 2) such as the RdNBR, are not direct measures of severity *per se*, but are useful for inferring severity when the fire effect(s) of interest can be meaningfully interpreted from imagery (e.g. Miller *et al.* 2009). Much of the confusion associated with terminology noted by Keeley (2009) and Lentile *et al.* (2006) can be alleviated simply by clearly articulating two factors suggested by Jain *et al.* (2004): (1) the element or aspect of severity being assessed or inferred (Holden *et al.* 2007; Miller *et al.* 2009), and (2) the specific timing of the post-fire assessment.

We suggest that recording actual fire effects measurements, such as percentage tree mortality or pre- and post-fire live tree basal area in forested areas (Miller *et al.* 2009), or average diameter of the smallest remaining branches in shrublands, is preferable to collapsing these measures into an index like CBI. We recognise that there will always be utility in composite metrics like CBI and GeoCBI, but without specific, ecologically meaningful measurements, it will remain difficult to directly relate ordinal severity class values to specific ecological characteristics or fire effects. These measures can be summarised as CBI or GeoCBI *ex post facto*, as appropriate. Severity classifications based only on relationships to composite measures may have little predictive power to describe potential severity before a site actually burns.

Moving towards more ecologically based severity classifications will require major improvements in the measurement of the direct effects of wildland fires. Developing meaningful relationships between individual fire effects or composite severity metrics and the conditions before, during and after fire will require studies with detailed quantitative descriptions of pre-fire conditions, fire behaviour and the post-fire environment at different time periods. Novel methods for assessing pre-burn

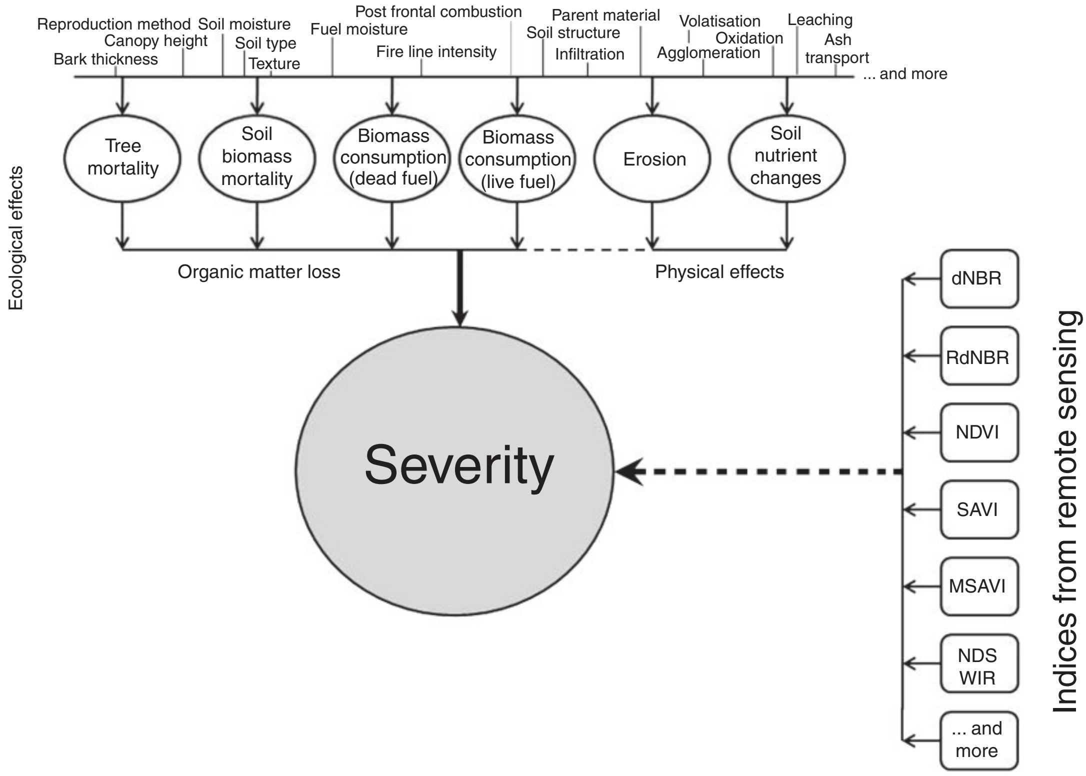

Fig. 2. Fire and burn severity, defined as the immediate and longer-term ecological effects of fire, can be assessed in the field using any one or a combination of metrics (top). Severity can also be inferred from individual remote sensing indices (side); this is only effective and interpretable when the index is correlated with fire effects on the ground.

conditions and for describing fire behaviour will be useful. Until we understand the causes and consequences of severity well enough to predict them, proactive, effective pre- and during- fire assessment and management to alter fire effects will continue to be challenging. Intensive spatial field surveys could support improved linkages between remotely sensed map products and field data, especially when fire effects vary greatly at fine spatial scales. One example of this in the US is the Accelerated Remeasurement and Evaluation of Burned Areas (or AREBA) project (Megown *et al.* 2011). The FIA program of the USDA Forest Service now measures both surface and crown fuels, along with many other ecosystem characteristics useful for quantifying fire effects, but the sparse distribution of plots established only in forested ecosystems will limit its operational use in severity mapping efforts, especially in rangelands.

Fire and burn severity mapping projects will continue to depend on remotely sensed imagery and field measurements. Therefore, it is critical that ecological advances in field assessments of severity be matched with the most appropriate imagery (Fig. 2). It is important that key fire effects are related to image signatures at appropriate scales. With the use of more advanced remote sensing technologies, such as hyperspectral imagery, LiDAR and radar, important fire effects may be more accurately and consistently inferred from imagery with higher spectral, spatial and temporal resolutions. It is exciting to see the many different research and management applications of severity, some of which have been prompted by the availability of MTBS data in the US. We look forward to learning as much about the causes and consequences of fire and burn severity as we know about fire behaviour. We also urge the development of a severity field assessment database and research to further our understanding of causal mechanisms linking fire and burn severity to conditions before and during fires to support improved models linking fire behaviour and severity and for forecasting effects of future fires. Understanding where, why and how fires burn severely will be greatly enhanced by efforts to: (1) relate severity to climate, weather, topography, fuels and land use (e.g. Dillon *et al.* 2011; Miller and Safford 2012), (2) explain temporal trends (Dillon *et al.* 2011;Miller *et al.* 2012; Mallek *et al.* 2013), and (3) develop tools that effectively link conditions before fire to flaming and glowing combustion, soil heating, biomass consumption and vegetation mortality. Better understanding will support better and proactive management of fire and fire effects.

One of the grand challenges for fire science remains to link conditions before, during and after fires together based on understanding of how fire behaviour causes fire and burn severity (Kremens *et al.* 2010). Without examining these linkages, it will be difficult to predict the ways in which pre-fire fuels and vegetation influences fire effects and vegetation response, yet that is key to proactive fuels and vegetation management. An important step towards meeting this challenge is a common base of terminology for severity that builds on measurable, physically based metrics linked to conditions before, during and after fires to characterise fire effects across multiple scales and applications. Only by taking this approach will the confusion and ambiguity be reduced and, more importantly, will our understanding of the ecological role of fire be enhanced. Ultimately, we need to more fully understand the causal mechanisms of severity, such as the multiple ecological interactions, scales of variability and fire behaviour drivers if we are to predict the consequences of alternative pre-, during and post-fire management strategies focused on influencing fire and burn severity outcomes.

Acknowledgements

We based this paper, in part, on the lessons we learned during several Joint Fire Sciences Program and National Fire Plan-funded projects that dealt with fire and assessing burn severity and creating severity classification systems. This research was supported in part by the USDA Forest Service Rocky Mountain Research Station, the University of Idaho (09-JV-11221637–270), the National Fire Plan and the Joint Fire Science Program (JFSP-09–1-07–4). We appreciate constructive reviews from Pete Robichaud and anonymous reviewers.

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