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Research Note

Towards a new paradigm in fire severity research using dose–response experiments

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Abstract. Most landscape-scale fire severity research relies on correlations between field measures of fire effects and relatively simple spectral reflectance indices that are not direct measures of heat output or changes in plant physiology. Although many authors have highlighted limitations of this approach and called for improved assessments of severity, others have suggested that the operational utility of such a simple approach makes it acceptable. An alternative pathway to evaluate fire severity that bridges fire combustion dynamics and ecophysiology via dose–response experiments is presented. We provide an illustrative example from a controlled nursery combustion laboratory experiment. In this example, severity is defined through changes in the ability of the plant to assimilate carbon at the leaf level. We also explore changes in the Differenced Normalised Differenced Vegetation Index (dNDVI) and the Differenced Normalised Burn Ratio (dNBR) as intermediate spectral indices. We demonstrate the potential of this methodology and propose dose–response metrics for quantifying severity in terms of carbon cycle processes.

Additional keywords: fire behaviour, fire effects, intensity.

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Introduction

As of 2015, over 2000 articles in the earth and environmental sciences contained the phrase *fire severity* and over 209 articles within just the current journal have used the term *severity*. This wide interest has resulted from studies that have reviewed terminology and quantification methods (Lentile *et al.* 2006*a*; French *et al.* 2008; Keeley 2009), inferred post-fire effects from field and remote sensing metrics (Ryan and Noste 1985; Miller and Thode 2007; French *et al.* 2008; Morgan *et al.* 2014) and modelled potential fire effects (Sikkink and Keane 2012). The prevailing approach to infer severity from wildland fires at landscape scales centres around the Differenced Normalised Burn Ratio (dNBR) spectral index and associated variants to broadly infer ecological change following fires (Morgan *et al.* 2001; Key and Benson 2006; Lentile *et al.* 2006*a*; Miller and Thode 2007; Lutz *et al.* 2011; Morgan *et al.* 2014). Rather than developing from scientific inquiry, these methods primarily evolved out of a need for management-oriented solutions in the US and elsewhere to assess post-fire sites for ecosystem rehabilitation and restoration (Eidenshink *et al.* 2007). Although these approaches are practical for managers, they have limited mechanistic and predictive power because they violate a range of principles (Roy *et al.* 2006; Lentile *et al.* 2009; Smith *et al.* 2010; Roy *et al.* 2013) as outlined below.

Post hoc ergo propter hoc fallacy ('after this, therefore because of this')

This is often described as a 'correlation implies causation' fallacy. Severity studies are often conducted in an opportunistic manner following unplanned wildland fires, where measurements could be made days or years following the fire. In most severity research, little to no pre-fire data are available and ecosystem change is often inferred from educated guesses about the pre-fire condition (many post-fire field observers have not visited locations before measurements). As described by Smith *et al.* (2010) and Roy *et al.* (2013) the temporal delays between each of pre-fire data (if available), active fire data and post-fire measurements can often lead to uncertainties as to whether the observed effects were caused by the fire and whether the magnitude of observed change can be solely attributed to the fire. These uncertainties are problematic because ecosystems are inherently dynamic: inter- and intra-annual variability in factors such as climate change, droughts, weather and population dynamics can create differences in species abundance, physiological performance and biogeochemical cycling, among other processes (Trenberth 2011; Moritz *et al.* 2012; Smith *et al.* 2014; Barbero *et al.* 2015; Kane *et al.* 2015*a*, *b*). Disturbances are regular components of ecological systems and are likely to occur at some scale between the fire event and sampling dates (e.g. land management activities, wind, drought and insects). Although satellite sensor imagery can be acquired near the time of the fire event, the use of such imagery to create differenced indices does not avoid the temporal separation between data points because more than a year can elapse between image acquisition dates. These extended dates are often used so that delayed mortality or ecosystem recovery can be evaluated, often at times of similar phenology to the pre-fire imagery. Therefore, although such imagery data selections are for the best of intentions, data collection after such extended times may limit the connections between cause and effect. For example, assessments of seedling counts acquired 5 years after a stand replacing fire may provide more information on seed dispersal mechanisms than they do on the effect of heat flux on seeds. The often opportunistic nature of fire severity research further compounds these issues because it can be difficult to find adequate areas to serve as unburned controls.

Severity definitions are not specific in a biological context and are not scalable

Many field metrics used in severity validation assessments exhibit limited mechanistic connections to ecosystem processes such as respiration, photosynthesis and net ecosystem productivity (Halofsky and Hibbs 2009). Of the various field metrics used to describe severity, many studies have highlighted that there is no consistent standard (Lentile *et al.* 2006*a*; Roy *et al.* 2006; Halofsky and Hibbs 2009). Although variability in the magnitude of fire effects is often documented, limited research exists that evaluates how increases may be caused by increasing 'doses' of fire behaviour metrics. There is an over reliance on statistical inference from fuel properties, with limited studies focusing on plant fire-mortality mechanisms (Van Wagner 1971; Michaletz and Johnson 2007; Michaletz *et al.* 2012). Carbon, water and energy fluxes (e.g. photosynthesis, transpiration) are all examples of physical base units that can be translated and scaled up to other ecological effects. Only a small minority of severity studies make direct connections to carbon stocks and fluxes (e.g. Conard *et al.* 2002; Kashian *et al.* 2006; Goetz *et al.* 2007; Hatten and Zabowski 2009; Romme *et al.* 2011).

Using data that are not objectively defined limits cross-comparisons

As described by Thomson (1889), the initiation of any physical science discipline first requires the identification of quantitative metrics that are unequivocally defined:

> When you can measure what you are speaking about, and express it in numbers, you know something about it; but when you cannot measure it, when you cannot express it in numbers, your knowledge is of a meagre and unsatisfactory kind: it may be the beginning of knowledge, but you have scarcely, in your thoughts, advanced to the stage of science, whatever the matter may be.

In fire severity assessment studies this is brought into sharp focus because the commonly applied dNBR imagery within the Monitoring Trends in Burn Severity (MTBS) product is simplified to subjective descriptors of *unburned*, *low*, *moderate* and *high* severity using arbitrary thresholds that often vary between fires from within the same ecoregion (Kolden *et al.* 2015). This results in maps that lack strong mechanistic connections to actual surface properties that have changed due to the fire. Equally, the commonly applied field Composite Burn Index is an aggregate of subjective ocular assessments of the likely ecosystem change (determined without pre-fire data) using categorical and binary data and is reported as a relative scale from 0 to 3 (Key and Benson 2006). Further to using the MTBS severity product, limited fire severity studies have used the continuous dNBR data to develop either regressions with surface change metrics (Lentile *et al.* 2009) or thematic maps of severity using ecologically based thresholds (Lentile *et al.* 2006*b*; Hall *et al.* 2008; Cansler and McKenzie 2012). Use of regressions or thematic maps that are calibrated to quantitative ground data helps overcome these cross-comparison challenges.

Ecological fallacy

In many cases correlations derived from aggregated populations (e.g. fire-, stand- and arguably even pixel-level data) do not necessarily translate to the same correlations applied to individual organisms (Schwartz 1994). In fire ecology, as in any other biological science, this highlights the need to conduct multi-scale studies that test and characterise such relationships. Landscape-scale severity assessments between spectral indices and field effects generally assume that field effects are aggregated to the scale of a satellite sensor pixel (usually 30×30-m pixels associated with the Landsat series), although few studies aggregate to multiple adjacent pixels to overcome locational uncertainty (Hudak *et al.* 2007). Measurements should be conducted at the scale relevant to the organisms or system being investigated, which may lead to challenges if seeking to assess mortality on individual trees with imagery more suited to plot- or area-based assessments. These challenges are confounded given plots are often correlated with imagery that incorporates a much larger spatial aggregation of post-fire effects and vegetation structure (i.e. mixed pixels). This is highlighted by the non-linear relationships between field measures of severity and imagery that can vary by spatial scale, ecosystem and soil type (van Wagtendonk *et al.* 2004; Lentile *et al.* 2006*a*; Hall *et al.* 2008; Smith *et al.* 2010).

As a result, it can be readily argued that severity research has to date been conducted in a manner that critically limits its utility in broader science inquiry and biophysical modelling. Overcoming this limitation will require a new paradigm for fire severity research.

Physics-based physiological response

Here, we present one such alternative pathway to evaluate fire severity by first reconsidering the problem from first principles. The main features of any scientific study evaluating responses of a system (i.e. plant to community scale) to the application of increasing quantities of an external stressor (fire in this case) are that both the dose and response metrics have units, are broadly transferable across different systems (i.e. fire regimes) and are readily scalable (Kremens *et al.* 2010). Further, we contend that these experiments should exhibit the following characteristics:

i. *Controls not subjected to the fire* to help decouple feedbacks and interactions. In field experiments these could be similarly instrumented paired watersheds as are commonly used in ecosystem science or fire hydrology research (Moody and Martin 2001), whereas in laboratory experiments these could be additional plants grown under the same conditions that are not burned;
ii. *Coincident measurements* to help reduce the potential for cause and effect fallacies and uncertainties by limiting temporal and spatial disconnects and the opportunity for other disturbances to confound the response metrics;
iii. *Cross-comparable* to investigate the effect of doses that have units (i.e. not an index, ratio or percentage), that are linked to quantifiable plant characteristics and are readily scalable. Spectral indices do have utility – especially in other applications of remote sensing – but should be unequivocally related to an actual surface metric that has units. It is important to note that spectral indices are inherently a function of the various surface interactions that can occur with sub-pixel objects (including scattering of linear and intimate mixtures) and thus surface metrics should ideally account for such mixtures (e.g. Smith *et al.* 2005) or represent measures aggregated to scales comparable to the satellite sensor pixels (e.g. Hudak *et al.* 2007);
iv. *Replications* to conduct the same treatments on multiple plants to reduce standard errors. Analysis should not hide the variation; rather analysis of variance should assess class means differences, whereas correlations and regressions should use the entire population to avoid ecological fallacies; and
v. *Mechanistic* to be broadly transferable across fire regimes. Dose and response metrics should be related respectively to quantitative heat transfer mechanisms and plant physiology processes. The plant physiology metrics could then enable the severity to relate to radiative transfer, hydrological, biogeochemical or ecosystem processes.

This is possible through the lens of either (1) highly instrumented field-based prescribed fire experiments (e.g. Southern African Fire Atmosphere Regional Experiment – SAFARI: Swap *et al.* 2003; RxCADRE: Ottmar *et al.* 2015; or the future Arctic–Boreal Vulnerability Experiment – ABoVE: http://above.nasa.gov/) or, as we describe in this paper (2) live plants subjected to laboratory combustion experiments using near-to-natural fuel beds (Fig. 1). Such experiments have been reported in the literature (e.g. Trollope *et al.* 1996; Jones *et al.* 2006; Frankman *et al.* 2013) – albeit not for the purpose of assessing fire severity – and represent the intersection of the *indoor* and *outdoor* fire science research as described by Van Wagner (1971). In the context of fire severity research, this separation is also apparent and is outlined in Table 1. A broad challenge is that specific fire effects are often a function of heterogeneous vegetation assemblages and structure, microclimate and local weather conditions at the time of the fire, and topography that all interact to produce unique conditions that are not readily achieved in laboratory or prescribed fire experiments. However, an advantage of laboratory experiments is that an experimental physics approach can be conducted where individual parameters can be held constant and then one parameter modified and its sensitivity evaluated. It can be readily argued that most contemporary severity research seeks to develop correlations between spectral indices and various field fire effects observations in the absence of a discussion of the heat transfer process (Morgan *et al.* 2014). A limited number of studies have focussed on heat transfer experiments to assess tree mortality mechanisms (Michaletz and Johnson 2007; Butler and Dickinson 2010; Kavanagh *et al.* 2010; Michaletz *et al.* 2012). However, a common limitation of these studies is the use of artificial heat sources (radiant heaters, collars, blow torches, etc.) rather than the exposure of trees to more natural fire conditions that distribute the heat dose over large sections of the plant. Further to these two categories, a limited number of studies have sought to infer severity from modelling (Chuvieco *et al.* 2006; Disney *et al.* 2011; Sikkink and Keane 2012).

We contend that the dose metrics should be related to radiant heat flux density (MJ m^{-2}) applied to the plants and that the response metrics could be related to fire effects on the terrestrial carbon cycle in vegetation (e.g. net photosynthesis). If the heat dose is always applied over similar temporal and spatial scales, this could also include fire radiant energy (FRE, in MJ) or if time scales are highly variable this could include the FRE density normalised by the burn duration. To demonstrate the potential of such a framework, an exploratory example follows using FRE density and net photosynthesis as the dose and response metrics.

Methods

Differing fire behaviour treatments were produced by varying the quantity of FRE density (MJ m^{-2}) released from a well-characterised 1-m^2 fuelbed and the fire severity was defined as changes in net photosynthesis (μmol CO_2 m^{-2} s^{-1}) at 4 weeks after the fire. Changes in the Differenced Normalised Differenced Vegetation Index (dNDVI) and the Differenced Normalised Burn Ratio (dNBR) were used as example intermediate indices (dimensionless) as they are often evaluated at landscape scales (Fig. 2).

An important distinction is that such intermediate indices act as a bridge between the fire behaviour dose and the severity response metrics and *maps of these indices should not be described as severity* (Morgan *et al.* 2014). Consequently, severity should always be reported in the units of the response variable that is related to the ecosystem process of interest.

Fig. 1. Examples of highly instrumented outdoor (*a*, *b*) and indoor combustion laboratory experiments (*c*, *d*). (*a*) Experiment conducted to measure fire behaviour in a southern African savannah in 2001 (described in Smith *et al.* 2005); (*b*) Thermal radiometer towers deployed during mastication fires in 2014 (equipment description described in Kremens *et al.* 2012); (*c*) example ignition experiment following methods described by Finney *et al.* (2013); and (*d*) lodgepole pine (*Pinus contorta*) seedlings undergoing surface fires.

Table 1. Van Wagner's (1971) indoor and outdoor fire science research descriptions, adapted to include the assessment of post-fire effects

Indoor fire severity research characteristics:
Fire behaviour is sufficiently complex that it is impossible to predict effects from models without using experiments.
Given that it is critical to understand the fire spread mechanism and at smaller scales the heat transfer mechanisms on individual organisms, controlled laboratory experiments are essential to characterise the relevant variables.
All internal and external variables that affect fire behaviour and post-fire effects are identifiable.
Experimental laboratory fires will yield valid relationships that are scalable to natural outdoor fires.
Outdoor fire severity research characteristics:
As long as it is possible to infer meteorological data and fuel properties, it is not necessary to characterise the mechanisms of fire spread or at smaller scales the heat transfer mechanisms on individual organisms.
Statistical approaches can be used to infer the natural variability of meteorological and fuel properties.

Experimental design

We used 36 2.5-year-old lodgepole pine (*Pinus contorta*) seedlings that were grown in a forest nursery following Dumroese *et al.* (2009). For the combustion experiments, nine seedlings were set aside as a control group and three sets of nine (in well-separated groups of two, three and four) were burned on fuelbeds associated with different FRE density doses (~0.4, ~0.8 and ~1.2 MJ m^{-2}). In this example, FRE density and total FRE are equivalent, as 1-m^2 fuelbeds were combusted. FRE was determined by combusting 15 oven-dried (~0% moisture content) *P. monticola* 1-m^2 fuelbeds varying in load from 0.1 to 0.5 kg and regressing measured FRE (as determined using dual band thermometry: see Kremens *et al.* 2010; Smith *et al.* 2013) to the pre-fire fuel load per square metre (FRE = 2.679 × pre-fire fuel load), where mean consumption was 84 ± 6%. These radiant energy density doses were selected to correspond to the range of typical energy release component (ERC) values expected for surface fires in forested systems using data from the literature (Kremens *et al.* 2012) and the English to metric conversion, FRE density (MJ m^{-2}) = 0.014 ERC (BTU ft^{-2}), where BTU denotes British thermal units. FRE provides a readily scalable measure of energy exposure because (i) it is an integrative measure of the radiative energy flux during the entire combustion process, (ii) considerable research links laboratory, field and remote sensing FRE studies, and (ii) there are numerous published linear relationships between FRE and

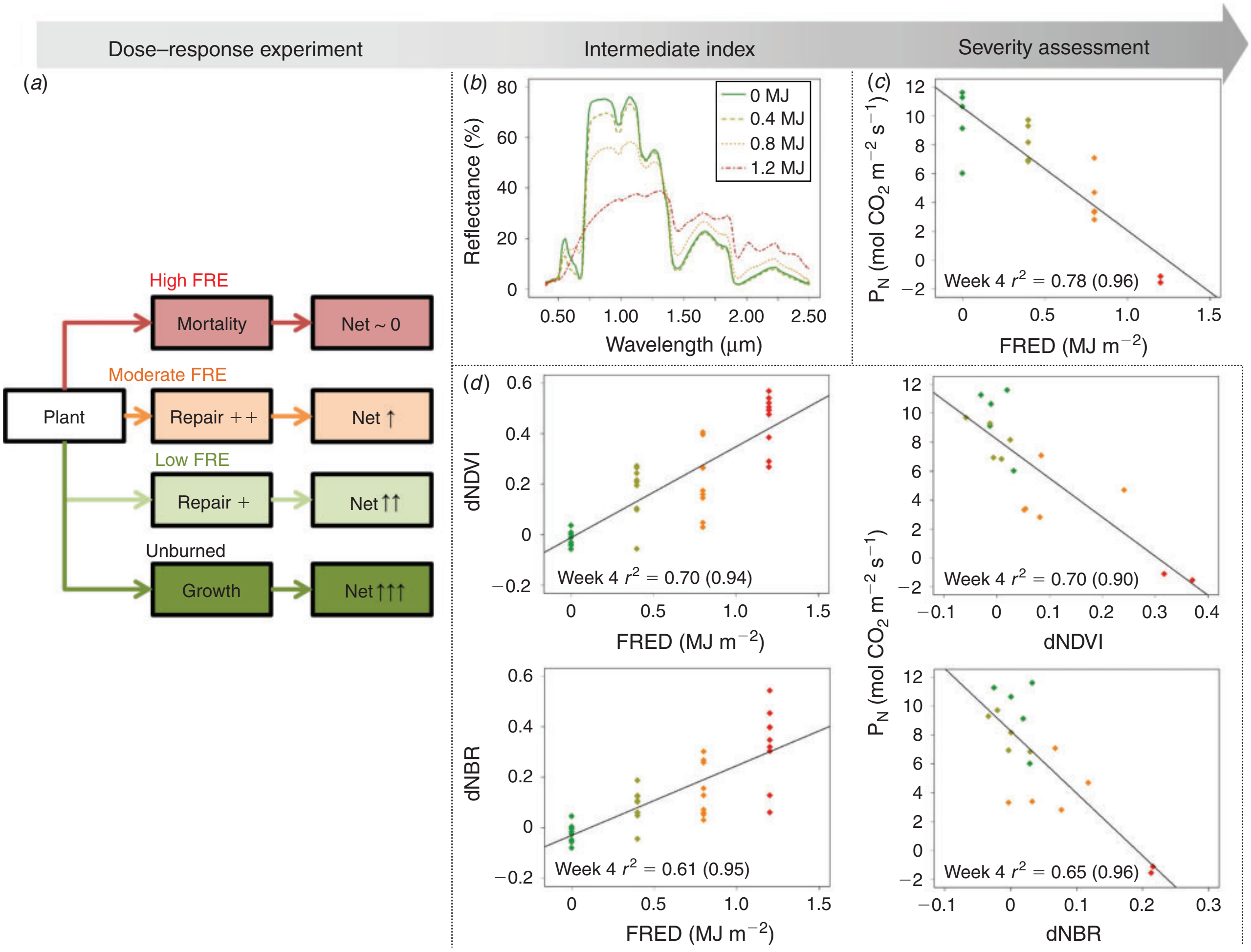

Fig. 2. Conceptual outline of an alternative fire severity pathway. (*a*) Hypothesis framework: as Fire Radiative Energy Density (FRED) dose increases, more of the plant's reserves will be used for repair leading to lower net photosynthesis. (*b*) Average spectral reflectance changes associated with FRED dose groups. (*c*) Direct observed dose–response relationship between FRED and net photosynthesis (P_N). (*d*) Intermediate dose–response relationships between FRED with differenced spectral indices and between differenced spectral indices and net photosynthesis. The reported r^2 values are for the entire population ($n = 36$ for spectra, $n = 17$ for net photosynthesis) and the r^2 of the means derived from each treatment group ($n = 4$) are shown in parentheses.

biomass consumption (e.g. Wooster *et al.* 2005; Kremens *et al.* 2012; Smith *et al.* 2013). Seedlings were returned to the greenhouse following the experiment. The experiment was a completely randomised design and pots were rearranged each week to minimise environmental variation associated with greenhouse bench position.

Fuelbeds and combustion

The fuelbeds consisted of oven-dried (~0% moisture content) *P. monticola* pine needles that were evenly distributed over 1-m^2 circular beds. This fuel type was used because its combustion properties have been well characterised. Each fuel bed was ignited via remote ignition using filament wire and a small quantity (1–2 g) of ethanol. These fires were combusted within a climate controlled combustion laboratory. The duration of the combustion was determined through detection of fire radiative power collected with a dual band thermal radiometer located above the experiment (Kremens *et al.* 2010; Smith *et al.* 2013) and across all the experimental burns a near constant combustion duration was observed ($\mu = 229$ s, standard error s.e. = 2.1 s).

Physiology and spectral experiments

Following combustion, the plants were returned to the nursery. Light-saturated (1100 mmol m^{-2} s^{-1} photosynthetic photon flux density; Glenn *et al.* 1984; Schoettle and Smith 1998) rate-of-photosynthesis measurements were performed on five random seedlings in each group using a LI-6400XT and 6400–05 conifer chamber (LI-COR, Inc. Lincoln, NE) and were expressed on a silhouette leaf-area basis. Identical measurements were also acquired before the combustion experiment. Spectral reflectance was collected from 1 week prior and at 4 weeks post-fire using a FieldSpec Pro spectroradiometer with the mineral probe attachment (Analytical Spectral Devices, Boulder, CO). Multiple (three to seven) pre-fire spectra were collected from both old (internodal) and new (apical bud)

Table 2. Mean and standard errors (shown in parentheses) of spectral indices and net photosynthesis data for different fire radiative energy density doses

FRED, fire radiative energy density; dNBR, Differenced Normalised Burn Ratio; dNDVI, Differenced Normalised Differenced Vegetation Index

FRED dose	Control	0.4 MJ m^{-2}	0.8 MJ m^{-2}	1.2 MJ m^{-2}
dNBR	−0.019 (0.015)	0.087 (0.043)	0.149 (0.050)	0.348 (0.051)
dNDVI	−0.016 (0.014)	0.172 (0.082)	0.201 (0.064)	0.457 (0.038)
Net photosynthesis	9.737 (0.763)	8.189 (0.441)	4.266 (0.572)	−1.326 (0.105)

foliage on each seedling by placing the probe on the plant, where each measurement averaged 10 collections from the radiometer. Post-fire spectra were collected from the same locations. Number of spectral measurements was dependent on availability of post-fire needles. The location of the new foliage spectra was coincident with net photosynthesis measurements. Between each seedling, a Spectralon panel calibration measurement was made to enable calculation of reflectance. All spectra were converted to band equivalent reflectance (Smith *et al.* 2005) associated with Landsat 8 for the calculation of dNBR and dNDVI.

Results

The means and standard deviations of the dNBR, dNDVI and net photosynthesis data are shown in Table 2. Fig. 2 shows that increasing FRE density dose on the treatment groups resulted in stepwise increases in dNDVI ($r^2 = 0.70$, s.e. $= 0.10$, $P < 0.001$) and dNBR ($r^2 = 0.61$, s.e. $= 0.10$, $P < 0.001$), with decreases in net photosynthesis ($r^2 = 0.78$, s.e. $= 1.92$, $P < 0.001$) at 4 weeks following the fire. The mean responses of net photosynthesis and dNDVI in each treatment group were significantly different from the control ($P < 0.05$; Tukey's honest significant differences test). The dNBR spectral index did not exhibit significant differences between the 0.4-MJ treatment group and the control. There were also significant differences between treatments in all cases, except between the 0.4- and 0.8-MJ treatments for dNDVI and dNBR. As the fire behaviour doses on the plants increase, the plants likely experience higher degrees of damage to leaf cellular structure that is then manifested through decreases in the near infrared spectral reflectance values and the associated dNDVI and net photosynthesis relationship ($r^2 = 0.70$, s.e. $= 2.25$, $P < 0.001$). The relationship between dNBR and net photosynthesis is lower ($r^2 = 0.65$, s.e. $= 2.40$, $P < 0.001$). Fig. 2 also highlights the erroneous results that can result from using regressions on means by showing the r^2 values from the means in parentheses.

Discussion

Within the methodology, we highlighted the link between unit area measures of ERC and FRE, where ERC is usually modelled and FRE is measured via field, aerial or satellite remote sensing. Because ERC is a predictive metric used by US land management agencies as part of the US National Fire Danger Rating System (NFDRS), using FRE would then improve the value of the NFDRS ERC data. ERC assessments are already being used in a predictive capacity to assess potential area burned in future fires (Freeborn *et al.* 2015). Potentially, linking FRE density and ERC with mechanistic knowledge of how varying heat doses affect tree physiology would be an important tool for researchers and land managers. However, before using FRE density as a 'severity' predictor, considerable research would be required to evaluate scaling of the controlled laboratory experiments to landscape-scale fires.

The *P. monticola* fuelbeds used in this example experiment were selected in an attempt to control the combustion characteristics across the different burns such that differences in FRE density dose on the physiology metrics could be elucidated. Clearly, fuel beds in the natural environment would be more complex and likely consist of heterogeneous fuel mixtures of varying moisture contents. The effects of such fuel mixtures (mixture of live and dead fuels, mixtures of different fuel types, mixture of fuels of different moisture contents, etc.) on the FRE released and other heat transfer metrics is a significant source of uncertainty but can be systematically tested within a series of controlled laboratory experiments (e.g. Viegas *et al.* 2010, 2013). Equally, broader transferability and scaling issues of such experiments should be considered before widespread application at landscape scales. For instance, research should assess whether similar results are apparent in other plant species or functional groups (shrubs, grasses, etc.) or how the observed relationships change with increasing tree size.

Although our example presented the dose and response metrics as FRE density and net photosynthesis, other metrics could readily be used. Clearly, the response units should be selected to match the scale and ecosystem process of interest: carbon cycle metrics when considering vegetation productivity, water cycle metrics when considering hydrology, etc. In the current study, FRE density was used as the dose metric because of its ease in repeatability. Caution should be applied using FRE density when large differences in duration are likely across experimental burns; in such cases the dose metric could be in terms of the heat flux incident on the plant or FRE density normalised by burn duration. Such metrics may be more appropriate in comparing results from different fires, experiments and fuel types where durations of energy flux incident on the plants may exhibit more variability. Future research should also compare fires of different durations to investigate the separate effects of energy dose and duration. Other dose metrics could include the conductive heat flux through the plant stem, the conductive heat flux through the soils to the plant roots or the convective heat flux on the plant canopy (Michaletz and Johnson 2007). The potential of convection is of particular note given that recent research highlights its importance in governing fire spread (Finney *et al.* 2015). Equally, research is needed to identify which of these heat transfer processes is primarily responsible for plant effects, perhaps by shielding the soils or canopy during experiments.

Conclusions

Here, we have highlighted an alternative experimental methodology to assess severity from wildland fires that could be performed within prescribed or laboratory fires. The main features of this proposed approach are that both the dose and response metrics have units and are broadly transferable across fire regimes, and that the research will ideally be readily scalable. The proposed framework could serve as an excellent starting point to advance fire severity research. Mechanistic fire effects research is critically needed to help us understand the influence of wildland fire on the global carbon cycle, forest productivity, and other direct and indirect effects on ecosystems. Such studies could help us better integrate wildland fire dynamics into biophysical and ecosystem process models. In summary, advancing the state of the science of fire severity will require that research is conducted in a systematic and quantitative manner similar to any toxicological dose–response experiments within the biological sciences, where the results are then characterised over a range of biological taxa and a series of spatial and temporal scales.

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