

The ability of winter grazing to reduce wildfire size and fire-induced plant mortality was not demonstrated: a comment on Davies *et al.* (2015)

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Abstract. A recent study by Davies *et al.* sought to test whether winter grazing could reduce wildfire size, fire behaviour and intensity metrics, and fire-induced plant mortality in shrub–grasslands. The authors concluded that ungrazed rangelands may experience fire-induced mortality of native perennial bunchgrasses. The authors also presented several statements regarding the benefits of winter grazing on post-fire plant community responses. However, we contend that the study by Davies *et al.* has underlying methodological errors, lacks data necessary to support their conclusions, and does not provide a thorough discussion on the effect of grazing on rangeland ecosystems. Importantly, Davies *et al.* presented no data on the post-fire mortality of the perennial bunchgrasses or on the changes in plant community composition following their experimental fires. Rather, Davies *et al.* inferred these conclusions based on their observed fire behaviour metrics of maximum temperature and a term described as the ‘heat load’. However, we contend that neither metric is appropriate for describing the heat flux impacts on plants. This lack of post-fire data, several methodological errors and the use of inappropriate thermal metrics limit the authors’ ability to support their stated conclusions.

Additional keywords: intensity, severity, thermocouples.

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Introduction

A recent study by Davies *et al.* (2015) sought to test whether winter grazing decreased wildfire size, fire behaviour and intensity metrics, and fire-induced plant mortality in shrub–grasslands in the western United States. In light of increasing wildfire size and frequency, as well as ongoing discussions regarding the listing of the greater sage-grouse as endangered in the United States, land-management options are needed to

reduce the negative effects of fire on sagebrush habitat and ecosystems. Although grazing may be an effective management tool in many circumstances, the study by Davies *et al.* (2015) does not demonstrate such benefits with respect to reducing wildfire sizes and fire-induced plant mortality in shrub–grasslands. We contend that the study by Davies *et al.* (2015) exhibits methodological errors, lacks data necessary to support the conclusions, and does not provide a thorough discussion of

the effects of grazing on plant communities. Specifically, (1) the inferred fire effects on plant mortality presented by the authors are predicated on the calculated term ‘heat load’, which is not a measure of heat; (2) the thermocouples used for their study are not adequate devices for the research objectives; (3) maximum temperature cannot reliably determine the heat transfer responsible for the fire effects on plant mortality; (4) methodological details were insufficient to replicate the study; (5) no post-fire effects or plant mortality data were presented to support the conclusions or the extrapolation of the results beyond the spatial and temporal scale of the experiment; and (6) the discussion was incomplete in considering the various effects of grazing on sagebrush ecosystems. We discuss several of these limitations in more detail in this commentary.

Methodological errors

Heat load is not a measure of heat

Use of the term ‘heat load’ by [Davies *et al.* \(2015\)](#) is incorrect because their measurement is duration of temperatures, not heat, and there is no way to calculate heat transfer from this measurement. This is most apparent in examining the units of measurement in which the authors report a time-integral of temperatures exceeding 60°C with units of degree-seconds rather than units of energy (e.g. joules, calories and British thermal units). It is worth noting that heat transfer is a function of temperature differences and not just temperature ([Wotton *et al.* \(2012\)](#)). Furthermore, heat load is a term already coined by the environmental biophysics community referencing the quantity of heat, not temperature, per unit time incident on a medium, to reach a specific outcome ([Campbell and Norman 2012](#)). Consequently, the use of the term ‘heat load’ by [Davies *et al.* \(2015\)](#) is inappropriate and misleading and their measurement has no reliable relationship to the resulting fire effects on plants.

The inadequacy of maximum temperatures

A key metric used by [Davies *et al.* \(2015\)](#) to describe the effects of grazing on fire behaviour was maximum temperature. However, maximum temperatures do not provide a strong mechanistic link to fire effects ([Van Wagner and Methven 1978](#); [Bova and Dickinson 2008](#)). The fire science literature has demonstrated that temperature-derived metrics are inadequate descriptors of fire intensity and that temperature might be a physically incorrect parameter because it is defined only for systems in thermodynamic equilibrium ([Van Wagner and Methven 1978](#); [Johnson and Miyanishi 1995](#); [Kremens *et al.* 2010](#); [Alexander and Cruz 2012](#)). When relating observed temperatures with fire intensity, the expected diffusion flame temperature in the burns of [Davies *et al.* \(2015\)](#) would typically all be within the same maximum range and of the order of 800–1100°C regardless of fire intensity ([McNaughton *et al.* 1998](#); [Smith *et al.* 2005](#); [Wotton *et al.* 2012](#)). Thus, the authors’ maximum temperature measurements are likely an overwhelming artefact of the thermocouple size (radiation, convection and response time) and their placement in air in the middle of the bunchgrasses, factors that the authors did not address. Consequently, the maximum temperature measurements have no reliable heat transfer relationship to the shrub–grass vegetation and thus no relationship to the resulting fire effects.

Inadequate thermocouple response times

[Davies *et al.* \(2015\)](#) do not provide information on the thermocouple time constant (Omega Engineering (Stamford, CT) has five different types of the applied thermocouples that exhibit time constants between 2 and 30 s in air). Time constants are the time taken to achieve 62.3% of any instantaneous temperature change in air moving at 19.8 m s⁻¹ (at room temperature and at atmospheric pressure). Consequently, whether 2 or 30 s, the thermocouple configuration described by [Davies *et al.* \(2015\)](#) cannot appropriately respond to the thermal transience of a flame front or the response characteristics of the fine dead and live vegetation. [Davies *et al.* \(2015\)](#) sampled air with these thermocouples at 1-s intervals. Without further discussion from the authors on how these data were averaged, it is impossible to determine data precision when using these devices. Given the slow response time of these thermocouples and the lack of discussion specific to signal averaging, they are unlikely to be an appropriate tool to assess the heat flux in this application.

The inadequacy of time-integrated temperatures

A detailed discussion of the correct use of thermocouples to assess heat impacts on vegetation can be found in the literature ([Bova and Dickinson 2008](#); [Wotton *et al.* 2012](#)). If [Davies *et al.* \(2015\)](#) sought to characterise the impact of the radiative heat flux on the plants, then time-integrated temperature or maximum temperatures are not valid proxies. The radiative heat transfer process is proportional to the fourth power of temperature, making a linear temperature with time product inadequate to describe this mechanism ([Van Wagner and Methven 1978](#); [Johnson and Miyanishi 1995](#); [Kremens *et al.* 2010](#); [Alexander and Cruz 2012](#); [Wotton *et al.* 2012](#)). Further complicating the assumption that their time-integrated temperatures can serve as proxy for heat flux measurements is the lack of description of the vegetative thermal boundary conditions. For the highly transient conditions of a flaming front, the thermal boundary conditions of the vegetative surfaces and the thermal diffusivity of the plant components determine the heat leading to potentially lethal temperatures. None of the temperature measurements of [Davies *et al.* \(2015\)](#) provides the information necessary to calculate plant temperatures and thus the potential for post-fire mortality.

Insufficient methodological details to replicate the experiment

[Davies *et al.* \(2015\)](#) included insufficient details regarding (1) how ignition was conducted, and (2) how specific fire behaviour characteristics were measured; and (3) they failed to provide any post-fire data beyond burned areas. For ignition, it is unclear whether drip-torch ignition occurred in or outside the plot and whether steady fire spread occurred by the time the fire fronts reached their experimental devices. Information is also lacking on the grazing history at the selected experimental locations and whether these locations were extensively grazed or representative of undisturbed ecosystems before the experiments. In addition, there is no evidence that the prescribed fire was ignited within the range of normal conditions for wildfire growth in the region. The authors imply that their prescribed fires occurred under wildfire conditions, and cite suppression activity on the Blitzen and Bone Creek Basin wildfires during

the timing of the prescribed fires. However, both of these short-duration, wind-driven wildfires were in mop-up phase and experienced zero growth for several days before the authors' prescribed fire (InciWeb 2015).

Davies *et al.* (2015) provide an inadequate explanation of how flame lengths, flame heights, rate of spread and flame zone depths were observed. Specifically, Davies *et al.* (2015) suggest they could measure flame height to a precision of 0.25 m, presumably from within the 10-m breaks on the outside of the 50 × 50-m plots. Adding further confusion to this measure is a lack of discussion on how they determined the 'top' of the flame (e.g. tallest vertical extent of flames vs highest point of the continuous flame). Rate of spread was calculated by timing the flaming front with a stopwatch as it travelled between stakes that were 2 m apart along the assumed direction of the prevailing winds. Similarly to the flame height discussion above, there was no discussion of specifically how flame zone depth was measured or estimated. The reader is informed that there are metal stakes on the ground separated by 2 m, but there is no further conversation as to other potential surface markers that would justify the reported 0.5-m precision in flaming depth.

Lacks data necessary to support the conclusions

No post-fire measurements presented

Davies *et al.* (2015) suggest that some of the primary implications of their research are that 'ungrazed areas may experience greater fire-induced mortality of native perennial bunchgrass' and that winter grazing 'increases the probability of post-fire recovery of the native plant community', 'decreases the probability of post-fire exotic annual grass invasions and subsequent development of an exotic annual grass-fire cycle' and 'reduce(s) the risk to sage-grouse and other sagebrush-associated wildlife'. Because no post-fire data were presented on perennial grass mortality, plant community composition or invasive species abundance, it is necessary that there be well-established mechanistic links between the measured fire behaviour metrics and the changes in these ecological metrics (Van Wagner 1971). In order to create such a link, Davies *et al.* (2015) made the statement that 'Higher temperatures and longer periods of elevated temperatures during a fire increase the likelihood of perennial grass mortality (Wright and Klemmedson 1965; Wright 1970; Odion and Davis 2000; Palaez *et al.* 2001)'. However, the studies of Wright and Klemmedson (1965) and Wright (1970) each used a mobile combustion chamber that does not represent natural burning conditions. These early studies used a simplistic set-up that does not reflect the current state of science for fire metrology and the fire science community's understanding of heat transfer impacts on vegetation (Michaletz and Johnson 2007; Butler and Dickinson 2010; Kremens *et al.* 2010; Tachajapong *et al.* 2014). The two other studies cited were from different ecosystems, where Odion and Davis (2000) focussed on heat impacts on seedbanks and Palaez *et al.* (2001) reused the mobile combustion chamber methodology in Argentina. Regardless, Wright and Klemmedson (1965) did not generally observe any significant differences in mortality 1 year post-fire between using the 93 and 204°C (soil temperature) treatments. Davies *et al.* (2015) relied on their maximum temperature data and heat load metric to infer impacts

on grass mortality. However, as we already outline above, neither of these fire behaviour metrics is appropriate.

Extrapolation beyond the scale of the research

We were also concerned by instances where the authors made important conclusions that cannot be directly supported by the presented data or by findings elsewhere in the peer-reviewed literature. In describing the maximum temperatures and heat loading at the microsites, Davies *et al.* (2015) concluded that 'Shrubs in ungrazed areas were more engaged by the fire; thus they burned more completely and released more energy (greater maximum temperature and heat loading)'. However, there were no measurements of the amount of shrub biomass consumed and no way of differentiating whether the higher recorded values of degree-seconds beneath the shrubs in the ungrazed plots were caused by more shrub combustion or by the combustion of more non-shrub fuels. Furthermore, as described above, energy was not measured.

Although Davies *et al.* (2015) acknowledge that under extreme weather events, the effects of winter grazing may be moderated, several of their discussion points overstate the potential of winter grazing under less than extreme weather conditions. For instance, although the controlled prescribed fires conducted in their experiment were fuel-driven, there is considerable evidence that most large rangeland fires are wind-driven events that move through shrub or grass crowns (Cheney *et al.* 1993; Cheney and Gould 1995; Launchbaugh *et al.* 2008). Several of the largest rangeland fires in recent years have burned the majority of their area in just a few days of progression, owing to wind-driven runs. Examples include the 2007 Milford Flat Fire, which burned 114 300 ha in just 48 h; the 2007 Murphy Complex, which burned over 80 000 ha in 2 days (Launchbaugh *et al.* 2008); the 2014 Carlton Complex, which burned 84 000 ha in 3 days; and the 2014 Buzzard Complex, which burned 146 400 ha in 3 days (InciWeb 2015). Launchbaugh *et al.* (2008) specifically found that grazing had little impact on fire spread during the 2007 Murphy Complex fire owing to the extreme conditions and winds that facilitated the extensive fire growth. Consequently, the following statements in Davies *et al.* (2015) are not supported by the presented data and are overstated: 'Wildfire suppression efforts may also be more effective in winter-grazed areas, further decreasing the probability of large fires' and 'Our research demonstrates that winter grazing is a pre-emptive treatment that can alter fire behaviour, area burned and fire intensity in at least some wildfires in *A. tridentata* subsp. *wyomingensis* communities and likely other shrub-grasslands'. Furthermore, there are no data presented to support the conclusion that reduced herbaceous fuel loads would produce smaller wildfires. Kolden *et al.* (2012) found only weak relationships between fire size and proportion of unburned area in forests; as far as we are aware, a comparable study has not yet been performed for shrublands and thus is a clear research need. However, given no studies have yet disproved the claim that winter grazing could decrease wildfire in some situations, additional research is clearly warranted that overcomes the methodological errors apparent in Davies *et al.* (2015). In re-investigating the utility of winter grazing to reduce fire intensity, future studies could build off prior research designs

that have conducted large grazing manipulation experiments in similar environments. For example, [Cheney *et al.* \(1993\)](#) reported that cut or grazed pastures led to fire spread rates ~20% lower than that observed in undisturbed sites.

No thorough discussion presented on the effects of grazing on plant communities

The scientific debate on the potential interactions between grazing and invasive species is complicated by factors such as nutrient and water availability, existing plant community resiliency, soils, climate, genetics and spatial scale ([Fleischner 1994](#); [Knapp 1996](#); [DiTomaso 2000](#); [Safford and Harrison 2001](#); [Adler *et al.* 2005](#); [Chambers *et al.* 2007](#); [Loeser *et al.* 2007](#); [Veblen *et al.* 2015](#)). Many studies have noted that grazing increases the establishment of invasive species through mechanisms such as soil disturbance and preferential consumption of certain species by livestock, among a wide variety of other processes ([D'Antonio and Vitousek 1992](#); [Chambers *et al.* 2007](#); [Reisner *et al.* 2013](#)). Likewise, in some studies, the focussed application of grazing has had no effect ([Stohlgren *et al.* 1999](#)) and in other studies, has demonstrated reductions in the quantity of cheatgrass ([Mosley 1996](#); [Diamond *et al.* 2009](#)). After describing the results of a related study where long-term grazing during the growing season decreased exotic annual grass cover after fire ([Davies *et al.* 2009](#)), [Davies *et al.* \(2015\)](#) concluded that 'winter grazing likely decreases the probability of post-fire exotic annual grass invasion and subsequent development of an exotic annual grass–fire cycle and increases the probability of post-fire recovery of the native plant community'. However, although elsewhere in the paper, it had been acknowledged that grazing can serve as a vector for invasive annual grasses, research implicating grazing as a factor increasing dominance of invasive annual grasses in the Great Basin was ignored here ([DiTomaso 2000](#); [Chambers *et al.* 2007](#); [Reisner *et al.* 2013](#)). In addition, no evidence has been shown in the wider literature or the current study that winter grazing can break the annual grass–fire cycle or assist in post-fire plant community recovery. Given the persistent cheatgrass seedbank, understanding the effects of winter grazing on cheatgrass reproduction and growth and native plant community dynamics is necessary to determine if winter grazing would decrease fire risk in the long term by altering seed-bank dynamics.

Perhaps the strongest statement by [Davies *et al.* \(2015\)](#) is that 'Clearly, a strategic landscape vision needs to be developed to apply winter grazing to reduce the risk of wildfire in sagebrush habitat and simultaneously maintain a wide diversity of habitats including habitats with enough residual vegetation for nesting sage-grouse'. However, we do not yet see sufficient empirical evidence to support these conclusions. Grazing can have a variety of effects on the landscape beyond influencing fire regimes, including changes in plant species composition, nutrient cycling, biological soil crust, soil erosion and animal diversity – some of which negatively impact ecosystems ([Weltz *et al.* 1998](#); [Jones 2000](#); [Greenwood and McKenzie 2001](#); [Chambers *et al.* 2007](#); [Lunt *et al.* 2007](#)). [Davies *et al.* \(2015\)](#) did not address the potential risks associated with grazing or provide any data on landscape-level effects, including the effects of winter grazing on habitat diversity and habitat required by sage-grouse.

Conclusions

Although the present commentary highlights key scientific shortcomings in the study by [Davies *et al.* \(2015\)](#), winter grazing may have desirable effects on managing wildfire in shrub–grasslands and other ecosystems. For instance, reduction in fuel loads and increases in fuel discontinuity by winter grazing in a system not regularly grazed may reduce rate of spread in non-wind-driven fires. Such strategies could assist land managers in reducing the negative consequences of unwanted fire behaviour but likely only in specific (and perhaps limited) situations. However, the paper by [Davies *et al.* \(2015\)](#) is not able to assess such questions and make generalisations about the use of winter grazing to reduce fire behaviour and post-fire mortality given it contains methodological errors, uses inappropriate active fire behaviour metrics, and has a near absence of post-fire data. As a consequence, significantly more research is warranted to overcome these shortcomings and evaluate when and how to apply winter grazing to reduce fire behaviour metrics and fire-induced mortality.

Overall, the limitations of [Davies *et al.* \(2015\)](#) make it difficult to draw accurate conclusions about grazing effects on plant mortality, community composition or wildlife habitat. This makes it impossible to judge the legitimacy of many of the authors' discussion and conclusion statements. Although we highlight a series of limitations in this study, we believe this example serves as a lesson learned illustrating the need for interdisciplinary expertise in investigating the complex implications associated with managing fuel complexes, fire behaviour and resultant effects.

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