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Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests

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ABSTRACT

Forests and woodlands in the central Rocky Mountains span broad gradients in climate, elevation, and other environmental conditions, and therefore encompass a great diversity of species, ecosystem productivities, and fire regimes. The objectives of this review are: (1) to characterize the likely short- and longer-term effects of projected climate changes on fuel dynamics and fire regimes for four generalized forest types in the Rocky Mountain region; (2) to review how these changes are likely to affect carbon sequestration, water resources, air quality, and biodiversity; and (3) to assess the suitability of four different management alternatives to mitigate these effects and maintain forest ecosystem services. Current climate projections indicate that temperatures will increase in every season; forecasts for precipitation are less certain but suggest that the northern part of the region but not the southern part will experience higher annual precipitation. The increase in temperatures will result in a greater proportion of winter precipitation falling as rain, earlier spring snowmelt, and a consequential increase in the length and severity of fire seasons. Fire frequency is likely to increase in the short term in all areas because of the warmer, longer, and drier fire seasons, but this change is likely to lead to a longer-term reduction in vegetation productivity in some of the most moisture-limited forest types, such as piñon-juniper and lower montane. This will decrease fuel accumulation rates and consequently reduce fire risk and result in longer fire return intervals. We consider four main management alternatives: fire suppression, wildfire (no intervention), prescribed fire, and mechanical thinning. The paper summarizes the effects of these treatments on forest ecosystem services, showing that they vary widely by forest type. This broad-scale assessment provides general guidance for forest managers and policy makers, and identifies more specific research needs on how climate-driven changes in fuel production and forest conditions will affect impact the four main forest ecosystems across the central Rocky Mountain region.

1. Introduction

Forests and woodlands in the central Rocky Mountain region of Montana, Idaho, Wyoming, Colorado, and Utah encompass a broad diversity of species, ecosystem productivities, community structures, and fire regimes. As a consequence of this diversity, climate change is likely to impact the region's forested ecosystems in varied and complex ways. The objectives of this review are: (1) to characterize the likely short- and longer-term effects of projected climate changes on fuel dynamics and fire regimes for four generalized forest types in the Rocky Mountain region; (2) to review how these changes are likely to affect carbon sequestration, water resources, air quality, and biodiversity; and (3) to assess the suitability of four different management alternatives to mitigate these effects and maintain forest ecosystem services. In the first section, we describe the general patterns of species composition, community structure, fire regimes, and land management legacies for four broad forest types: piñon-juniper woodlands, lower montane forests, upper montane forests, and subalpine forests. We then provide a conceptual model relating fire regimes to climate. Next, we summarize climate forecasts for the region and use the conceptual model to project how fire regimes in the four forest types are likely to change in both the short and long term, as these two different time scales may show different trends. Finally, we compare four management alternatives – fire suppression, wildfire (no intervention), prescribed fire, and mechanical thinning – in terms of their costs and benefits for preserving or

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promoting carbon sequestration, water resources, air quality, and biodiversity preservation under possible future climate changes.

The Rocky Mountain region encompasses a complex geologic template with steep climatic and topographic gradients and diverse soil substrates. Superimposed on this template are the legacies driven by past climatic events, such as forest establishment pulses and natural disturbances such as insect attacks and fire. More recent legacy effects include land use change and land management practices, such as logging, grazing, and fire suppression. Past fire management practices can greatly complicate efforts to project ecosystem response to future climate and fire management alternatives. For some forest types fire exclusion has caused profound changes in forest composition, structure, type and amount of fuel, and resulting fire behavior. Dendroecological studies have provided abundant information on historical fire regimes, and these have helped define the historic ranges of variability in disturbance processes and forest conditions before widespread impacts due to Euro-American settlement. In many cases, forest structure has been altered to the point where longer-term fire regimes are no longer operable without major ecological restoration efforts (Friederici, 2003).

Given the broad goals of this paper, we necessarily have to group Rocky Mountain forests into four broadly defined forest types: piñon-juniper woodlands, lower montane forests, upper montane forests, and subalpine forests (Fig. 1). This classification allows us to provide an overall characterization, context, and contrast that is useful for forest managers, decision makers, and the public. By definition we cannot capture all of the spatial complexity and temporal history of forest ecosystems across the Rocky Mountain region (e.g. Rollins et al., 2002; Romme et al., 2009). However, these four broad forest types capture much of the variability among forests and fire regimes in the region because of the wide range of climatic conditions, forest structure, and forest productivities represented by these four main forest types. It also is important to recognize that a majority of the region is covered by non-forest ecosystems (Fig. 1). Some of the information and analysis in this report may apply to grasslands or shrublands adjacent to the forested ecosystems, but these ecosystems are not an explicit focus of this review. Other less extensive forest types, such as riparian forests, are not covered in this review, even though they may be especially vulnerable to climate change and fire regimes. The following sections describe and briefly outline the historical and current conditions for each of these four forest types.

1.1. Piñon-juniper woodlands

Piñon-juniper woodlands cover extensive areas in the southwestern US and Great Basin. Juniper woodlands without piñon also are common in dry areas of southern and central Wyoming. Piñon-juniper woodlands are characterized on the Colorado Plateau of southern Utah and southern Colorado by two-needle piñon (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*). Farther west the dominant species change to single-needle piñon (*Pinus monophylla*) and one-seed juniper (*Juniperus monosperma*), and these species cover large areas in southwestern Utah and certain mountain ranges in the basin-and-range province of western Utah. Piñon-juniper woodlands are typically characterized by relatively sparse, discontinuous understories of grasses, herbaceous plants, and shrubs, particularly the various subspecies of big sagebrush (*Artemisia tridentata*).

A recent review of piñon-juniper fire ecology and fire regimes described three main types of piñon-juniper woodlands, which reflect variations in soils, climate conditions, past disturbances, and climate (Romme et al., 2009). Many of the piñon-juniper woodlands in the central Rocky Mountain region are so-called "persistent woodlands," where soil and climate conditions are most favorable for piñon-juniper establishment and growth (Romme et al., 2009). In these persistent woodlands, canopies may be open or closed, and understory vegetation is typically sparse and discontinuous. Fires that burned across large areas were historically infrequent (ca. 300–500 year intervals) due to lack of surface fuels; these typically took the form of stand-replacing crown fires (Floyd et al., 2000), and probably required strong winds to burn across patchy fuelbeds. Rather than reflecting past fires, tree demography was likely determined by legacies of droughts and wet periods and, perhaps, on other disturbances such as bark beetles (Romme et al., 2009; Heyerdahl et al., 2011).

1.2. Lower montane forests

Lower montane forests in the Rocky Mountains are dominated by ponderosa pine (*Pinus ponderosa*). In the southern part of the region in Colorado and southeastern Utah, lower montane forests usually occur as either relatively pure stands of ponderosa pine or ponderosa pine with some combination of species found in lower-elevation piñon-juniper woodlands or higher-elevation upper montane forests, especially Douglas-fir (*Pseudotsuga menziesii*). Gambel oak (*Quercus gambellii*) also is a common understory species across the southern lower montane zone. Lower montane forests in Idaho and Montana consist of mostly ponderosa pine or are co-dominated by Douglas-fir or western larch (*Larix occidentalis*). Historically, lower montane forests also were characterized by diverse and productive understories, dominated by bunch grasses, although in many areas, increased tree density and canopy closure since fire exclusion has reduced the cover and diversity of understory vegetation.

Prior to Euro-American settlement, fire scars document surface fires every 3–30 years, depending on elevation, latitude, and environmental conditions (e.g. Agee, 1993; Falk et al., 2011). Mature ponderosa pine trees are well-adapted to surface fires, with thick bark that protects growing tissues from girdling and high crowns that lessen the possibility of crown scorch. Under the historical fire regime, larger and older trees tended to survive most fires, whereas seedlings and smaller saplings were usually killed. Crown fires were mostly limited to small patches. These passive crown fires, where fire spread was predominately through surface fuels, added to landscape diversity by creating meadows and openings. The mix of recurrent surface fires, passive crown fires, seedling and sapling mortality, and occasional sapling survival resulted in a diverse landscape mosaic of generally multi-aged, multi-sized, and mainly low- to moderate-density open-canopy forests (Friederici, 2003; Heyerdahl et al., 2011).

1.3. Upper montane forests

Upper montane forests in the Rocky Mountain region are cooler and wetter, and typically include diverse mixes of several tree species, although they also can be co-dominated by ponderosa pine and Douglas-fir. Common species include firs (*Abies* sp.), aspen (*Populus tremuloides*), and western hemlock (*Tsuga heterophylla*). Understories in the upper montane can be quite diverse, ranging from a near continuous mix of grasses, shrubs, and herbs to very sparse cover under relatively dense closed forest canopies.

Fire regimes in the upper montane were historically very heterogeneous (Agee, 1993; Grissino-Mayer et al., 2004; Schoennagel et al., 2004; Sherriff and Veblen, 2008; Schoennagel et al., 2011). Upper montane forests experienced a gradient from predominately surface fires at lower elevations and in drier aspects to a mix of surface fire, patchy crown fire, and large areas of active crown fire as elevation and moisture increased. Fire frequency patterns followed severity patterns, with high fire frequency and typically lower fire severities near the transition to lower montane forests and

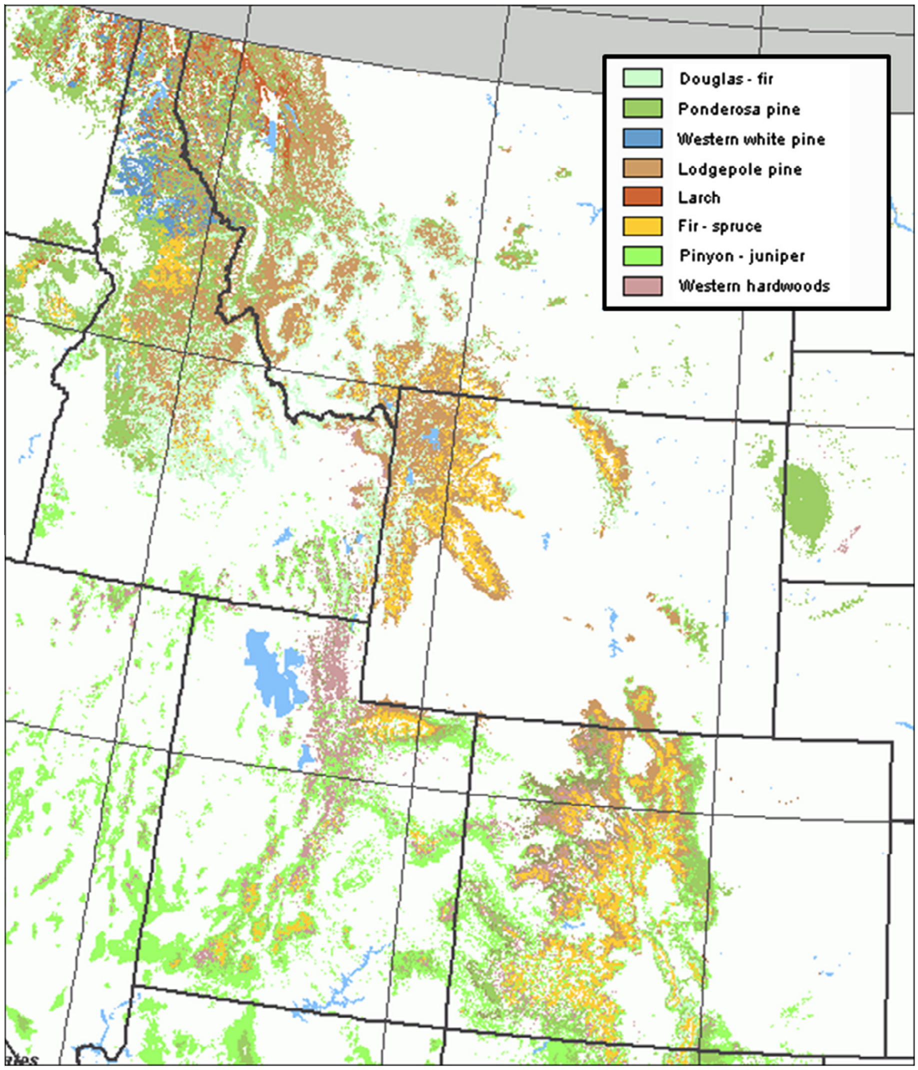

Fig. 1. Distribution of the dominant tree species in the Rocky Mountain region. For simplicity these are grouped into four broadly defined forest types: piñon-juniper woodlands, lower montane forests (primarily forests dominated by ponderosa pine), upper montane forests (primarily forests dominated by Douglas fir or diverse mixtures of species), and subalpine forests (forests dominated by lodgepole pine, spruce, fir, western white pine, and some hardwoods such as aspen). Data are forest cover types from the National Atlas (nationalatlas.gov).

decreasing fire frequency and increasing fire severity towards the subalpine (Baker et al., 2007; Sherriff and Veblen, 2007). This variability in fire frequency and severity, in combination with the variability in the physical environment, further contributed to highly diverse landscapes, including woodlands with widely spaced trees, dense closed-canopy forests, large open meadows, and a tendency towards even-aged forests on north-facing slopes and at higher elevations where stand-replacing fires were the most common.

1.4. Subalpine forests

Subalpine forests are dominated by Engelmann spruce (*Picea engelmannii*) and fir species in more productive sites and lodgepole pine in drier sites. Aspen also is present in many warmer subalpine forests. Five-needled pine species can be locally dominant, particularly in the driest, least productive sites, although they can form extensive landscapes of almost pure single-species forests in some areas. In most places, however, these five-needle pines form mixes with lodgepole pine and/or spruce-fir. Species include limber pine (*P. flexilis*), whitebark pine (*Pinus albicaulis*) in Idaho and Montana, and Rocky Mountain bristlecone pine (*P. aristata*) in southern Colorado and Utah.

Fires in subalpine forests were generally relatively rare, only occurring once every 100 to 400+ years, and were related to some of the strongest regional drought years (e.g. Schoennagel et al., 2005; Sibold and Veblen 2006). Temperature also was a factor, with warmer summers and early snowmelt related to regional fire years during the 20th century (Morgan et al., 2008). Forests are generally closed-canopy, so they usually burned as severe crown fires that killed most or all trees over hundreds to thousands of hectares (Sibold et al., 2006). Subalpine tree species tend to be well adapted to stand-replacing wildfire, as lodgepole pine regenerates from seeds that are released from serotinous cones, aspen sprouts new shoots from underground dormant buds, and spruce and fir seeds are winged to assist with long-distance dispersal.

2. Conceptual model of fuel dynamics and fire regimes

Fire regime refers to the typical fire frequency, general fire behavior, average fire sizes, and average season of burning. Fire regime is controlled primarily by the amount, type, arrangement, and condition (i.e., dryness) of fuels. Fuel characteristics are closely connected to climate, but on two different time scales (Hessl, 2011). On time scales of hours to months, variables such as relative

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humidity, temperature, wind speed, timing of snowmelt, and precipitation events affect fuel moisture (Goldammer and Price, 1998). Fuels have to be dry enough to burn, and changes in daily to seasonal weather conditions are the primary controls on whether ignition can occur and how extensive or intense a fire will be. However, climate affects fuel amounts on time scales of years to centuries by controlling species composition, ecosystem productivity, and rates of biomass decomposition (Meyn et al., 2007; Krawchuk and Moritz, 2011). The horizontal and vertical fuel structure greatly affects fuel connectivity across landscapes and the ability of fire to transition from surface to canopy layers.

The balance of fuel drying over shorter time scales and fuel production over longer times largely controls the general patterns of fire regimes and fuel dynamics across climate gradients (Fig. 2; Meyn et al., 2007; Hessl, 2011; Heyerdahl et al., 2011; Krawchuk and Moritz, 2011). In areas with warm and dry climates and relatively low net ecosystem productivity, fire regimes are generally limited by fuel amount. Fuels are sparse and patchily distributed, fire frequency is generally low, and when ignitions occur, fire extent tends to be limited. In such areas, which in the Rocky Mountain region include piñon-juniper woodlands, rare landscape-scale fires are typically driven by high winds that push flames across fuel gaps and ignite tree canopies, resulting in widespread tree mortality (e.g., Floyd et al., 2000).

In contrast, forests in cool and wet sites, such as the subalpine forests, tend to be relatively productive with higher fuel loads and connectivity. However, the fuels in these forests are rarely dry enough for extensive burning to occur, so fire frequency is relatively low (Fig. 2). When fires occur, they tend to burn at high severity, and the resulting active crown fires cause extensive areas of tree mortality (Schoennagel et al., 2004).

Between these two extremes of fuel-limited vs. climate-limited forests are ecosystems that are both fairly productive and that dry out often enough to make fires relatively common (Meyn et al., 2007; Krawchuk and Moritz, 2011). In the Rocky Mountain region, the lower montane and to a lesser extent the upper montane forests typically experienced relatively frequent, lower severity surface fires or passive crown fires (Fig. 2; Brown et al., 1999; Sherriff and Veblen, 2008). It is these fire regimes where a century of fire exclusion has had the most impact, with fuel load and forest structure profoundly altered from historical patterns (Agee, 1993; Schoennagel et al., 2004). Many present-day montane forests have greatly increased tree densities, lowered crown base heights, and lost landscape mosaics of openings, meadows, and variability in stand structures (Agee, 1998).

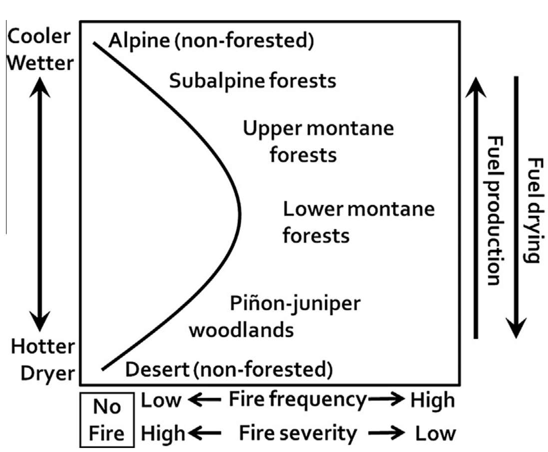

Fig. 2. Generalized fire regimes, defined by frequency and severity (curved line) across climatic gradients for the four forest types and non-forest ecosystems in the Rocky Mountain region. Fuel production and fuel drying are also shown in relation to the climatic axis, and this shows that fire regimes in drier and warmer ecosystems are typically "fuel limited", while fires in cooler and wetter ecosystems are typically "climate limited". Fire frequency and severity are highest in the lower montane ecosystems because they have the optimal combination of fuel production and fuel drying.

3. Climate projections and future fire regimes

This section summarizes projected climate change impacts on fuel dynamics and fire regimes over approximately the next 50 years in the central Rocky Mountain region. Downscaled climate change projections are drawn from Liu et al. (2013) (see also Rocca et al. this issue) and presented in comparison to other studies, conducted mostly at coarser spatial resolutions, that capture a variety of emissions scenarios, climate models, and forecasting methods.

Over the next several decades temperatures are forecast to increase in every season, with average increases of more than 3 °C in summer and 2–3.5 °C in the fall (Liu et al., 2013). Fire seasons generally occur from late spring to early summer in the southern Rockies, transitioning gradually towards late summer and early fall with increasing latitude (Mote et al., 2005; Westerling et al., 2006). The projected average temperature increases of 2–3 °C in winter and spring are likely to result in more of the winter precipitation falling as rain rather than snow (McCabe and Wolock, 2010). Warmer spring temperatures will result in earlier spring snowmelt and an earlier start to fire seasons. Warmer summers and falls will lead to a temporal extension of the fire season. The observed changes in temperature for forested areas in the central and northern Rocky Mountains are already having a profound effect on the timing of runoff in snow-dominated areas, with peak runoff commonly occurring two to three weeks earlier than previous decades (Cayan et al., 2001; Westerling et al., 2006; Clow, 2010). Therefore, not only will fire seasons start earlier and end later, but also they will likely be drier due to the earlier snowmelt and the direct impacts of the temperature increases on fuel dryness (Westerling et al., 2006; Morgan et al., 2008).

Region-wide, precipitation forecasts generally show an increase in winter precipitation and a decrease in summer precipitation (Liu et al., 2013), although there is less certainty in precipitation forecasts from the various models (Meehl et al., 2007; Ray et al., 2008; National Research Council, 2011; Notaro et al., 2012). Most forecasting efforts and ensemble models, such as the World Climate Research Programme Multi-model Dataset, show annual precipitation decreasing to the south of Colorado and Utah and increasing to the north, with a zone of uncertainty in between (Meehl et al., 2007; Gutzler and Robbins, 2011). The recent projection from Liu et al. (2013) shows summer rainfall decreases on the order of at least 50 mm, or approximately 25%, for Colorado. Fall, winter, and spring increases in precipitation are most pronounced in the northern Rockies of Idaho and Montana (Gutzler and Robbins, 2011; Liu et al., 2013).

There is nearly universal agreement that the projected changes in temperature and precipitation will increase the moisture deficit across the Rocky Mountains, even where precipitation is forecast to increase. Warming and drought will be most severe in the southern half of the region (Burke et al., 2006; Hoerling and Eischeid, 2007; Seager et al., 2007; Spracklen et al., 2009; Gutzler and Robbins, 2011). Gutzler and Robbins (2011) forecast Palmer Drought Severity Index (PDSI) across the western U.S. using an ensemble of global climate models with a mid-range (A1B) forcing scenario, empirically downscaled spatially and temporally. Their models show lower average PDSI (i.e., more droughty) conditions throughout the region, with especially severe drying in Colorado. Liu et al. (2010, 2013) use dynamically downscaled projections of the Hadley Center Climate Model (HadCM3) under the A2 emissions

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scenario, a relatively high emissions scenario. They forecast an increase in the Keetch-Byran drought index (KBDI; Keetch and Byram, 1968, revised 1988) for the vast majority of the Rocky Mountains, with summer and fall KBDI increasing by more than100 points across most of Utah, Colorado, Wyoming and Montana. For reference, summer KBDI between 1971 and 2000 averaged around 400 for Utah and Colorado, and around 300 for Idaho, Montana, and Wyoming; values greater than 400 are considered high fire potential, and above 600 are considered extreme (Liu et al., 2013). In much of Idaho, however, the increase in winter precipitation partially compensates for warmer temperatures, resulting in a KDBI increase of less than 50 points in summer and fall, and even some decreases in fall KBDI (Liu et al., 2013).

Several modeling efforts have explored the consequences of these moisture deficit projections for fire regimes in the Rocky Mountain Region. These studies have predicted increases in area burned (Bachelet et al., 2003; Spracklen et al., 2009; National Research Council, 2011), number of high fire danger days (Brown et al., 2004), and fire frequency (Krawchuk et al., 2009; Westerling et al., 2011) by the mid to late 21st century. A recent National Research Council Report (NRC, 2011) predicted that a 1 °C increase in global average temperature would result in a sixfold increase in annual area burned for the Southern Rockies (Colorado, Utah, and Wyoming), a fivefold increase in the Central Rockies (central Idaho and Montana), a fourfold increase for the Colorado Plateau, and a doubling in the Northern Rockies (northern Idaho and Montana). The predicted doubling of area burned in the Northern Rockies demonstrates the dominance of moisture deficit over the projected increase in precipitation for controlling fire regimes.

In these fire regime projections, fire occurrences are generally modeled using statistical relationships between historical (seasonal or annual) climate and past fire size or occurrence. Only the regional (Bachelet et al., 2003) and global-scale (Krawchuk et al., 2009) efforts using dynamic global vegetation models attempt to account for the trade-offs between fuel production and fuel condition, and these coarse-resolution efforts cannot capture the specific dynamics of climate change with forest type that have to be known when making land and fuels management decisions. In the absence of any mechanistic fire spread modeling studies like (e.g. de Groot et al., 2013) or landscape-scale forest dynamics modeling efforts (e.g. Miller and Urban, 1999; Loudermilk et al., 2013) on fire-vegetation-climate change interactions in the Rocky Mountain region, we take advantage of the conceptual model described above (Section 2) to inform our projections of future alterations to fire regimes.

4. Future fire regimes in four forest types

The conceptual model of fuel dynamics and fire regimes, when combined with the literature on future climates, allows us to project how climate change is likely to alter both short- and long-term fire regimes in each of the four forest types. The general patterns projected here then have to be adjusted according to local conditions, but climate forcing explains many of the overall trends in regional fire occurrence (e.g. Kitzberger et al., 2007; Krawchuk and Moritz, 2011). From the projected regional trends, we also can make inferences about changes in carbon sequestration, water quality, air quality, and biodiversity.

Our assessments are guided by the previously described climate forecasts, which cover approximately the next 50 years. We define "short-term" as the characteristics of the next fire to occur in an area, while "longer-term" refers to the trajectory of fire regimes over several fire cycles based primarily on changes in longer-term fuel characteristics (e.g., stand productivities). By definition, the short-term projections are based on existing fuel structures, some of which have deviated from historical conditions, while the longer-term projections have to balance the direct climatic effects with the indirect effects in terms of changes in vegetation, productivity, and fuel loads. These longer-term projections may only be manifested beyond the approximate 50-year time frame for the climate projections, especially for the forest types with long fire recurrence intervals, but the strength of the basic trends and our understanding give us confidence in the likely fire regime and how this will affect the other resources being considered here.

We define fire risk as the probability of fire occurrence, with the implication that over longer time scales fire risk relates to fire frequency. We define fire severity as the degree of organic matter loss in the fire (sensu Keeley, 2009). According to this definition, a high severity fire is one in which most or all of the aboveground vegetation is killed, and virtually all of the litter and organic matter at and above the soil surface is consumed. In contrast, a low severity fire would not kill all of the aboveground live vegetation, particularly the larger trees, and there would still be some litter to protect the soil surface and no loss of soil organic matter. In both our short- and longer-term forecasts we assume no change in forest type or dominant tree species, which is generally reasonable given the time frame of this assessment. We recognize, however, that ecosystem type conversions, such as changes from forest to shrubland or grassland, may take place due to climate change or uncharacteristic fire regimes (e.g. see Westerling et al., 2011). The possibility of such type conversions is discussed in the context of management alternatives (Section 5).

In the short term, warming and/or drying will increase fuel drying and lead to a higher probability of ignition and burning across all forest types. The overall result is increased fire occurrence, more extensive areas of burning, and in some cases increased fire severity (Table 1). Over the longer term, however, future warming and drying will lead to altered fire regimes by affecting net ecosystem production (Krawchuk and Moritz, 2011). Lower fuel production in the drier ecosystems may lead to decreased fuel loads and loss of fuel connectivity across landscapes and, in these areas, climate change may actually contribute to reduced fire frequency (Swetnam and Betancourt, 1998; Westerling and Bryant, 2008; Littell et al., 2009). In these cases, an initial, potentially higher severity fire may reset the ecosystem by consuming existing fuels and modifying fuel structures. We then postulate a subsequent shift downwards towards the hotter and drier conditions in Fig. 2 (Table 1).

The confidence level is listed for each our fire regime projections in Table 1 using the National Climate Assessment guidelines (National Research Council, 2011). High confidence indicates strong evidence and high consensus; medium confidence indicates moderate or suggestive evidence; and low confidence indicates inconclusive evidence, extrapolations, and/or lack of opinion among experts. We generally have more confidence in our short-term projections, and this is due in part to our greater knowledge of current fuel loads and hence our projections for the next fire. We also have an increasing number of observations on climate change-induced alterations to fire regimes (Floyd et al., 2004; Westerling et al., 2006; Littell et al., 2009). Our confidence in our longer-term projections is only medium at best due to the uncertainty in: (1) the rate at which changes in climate will affect vegetation productivity relative to our time horizon of 50 years; (2) the validity of using or extrapolating past fire-climate relationships for future, possibly non-analog climate conditions (Williams and Jackson, 2007); (3) how a warmer and drier climate will alter ecosystem productivity in the upper montane and subalpine forests; (4) complicating factors such as insect outbreaks and invasive species that can change fuel structures and alter fire regimes; (5) uncertainty about future land use and land management practices; and (6)

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Table 1
Forecasted change in fire risk (probability of fire occurrence) and fire severity (effects of fire on forest overstory) over the short and longer term by forest type. + indicates an increase in fire risk or severity, 0 indicates no change, and – indicates a decrease. Short-term projections assume current fuel structures and represent our projections for the next fire in an area. Long-term trajectories consider recovery from the first fire and the likely fuel production over one or more fire cycles. A level of confidence is associated with each projection, and these recognize the numerous, non-climate complicating factors.

	Short-term change in fire risk (likelihood of the next fire)	Short-term change in fire severity during the next fire	Longer-term trajectory for fire risk (fire frequency)	Longer-term trajectory for fire severity	Complicating factors
Piñon-juniper	+ (high confidence)	0 (high confidence)	– (low confidence)	0 (low confidence)	• Cheatgrass and other invasive grasses
Lower montane	+ (high confidence)	+ (high confidence)	– (low confidence)	+ (low confidence)	• Heterogeneity in management actions and restoration activities
Upper montane	+ (high confidence)	+ (high confidence)	+ (low confidence)	+/– (low confidence)	• Heterogeneity in management history and restoration activities
Subalpine	+ (high confidence)	0 (high confidence)	+ (medium confidence)	0 (medium confidence)	

changes in the frequency and location of fire ignitions, both human and natural. With respect to the latter, increased temperature and summer moisture deficits may increase lighting activity and fire starts (Price and Rind, 1994), but the current state of knowledge does not permit the explicit prediction of such changes (Hessl, 2011). Similarly, there is no way to reliably predict the changing potential for increased human ignitions.

4.1. Piñon-juniper woodlands

For piñon-juniper woodlands our short-term projection is for fire risk to increase. This projection has a high confidence level because an increased occurrence of extensive, high-severity fires in piñon-juniper have already been observed in response to recent warm, dry periods (Floyd et al., 2004). We expect no immediate change in fire severity because fires, at least the ones that burn a large area, already consume most of the available fuels and kill the tree overstory (Baker and Shinneman, 2004).

We project with lower confidence that the longer-term trajectory for current piñon-juniper landscapes will be less frequent fire (lower fire risk). Following an initial fire in any particular location, we expect that growth of surface fuels as well as trees to be reduced as a consequence of moisture limitation, resulting in a sparse (or absent) canopy and disconnected fuels. Fires that do occur will, by necessity, be associated with high winds, so we expect with low confidence that fires will remain stand replacing. Both the short- and long-term projections for piñon-juniper woodlands are complicated by the increased prevalence of invasive annual grasses, especially cheatgrass (*Bromus tectorum*). After cheatgrass-fueled fires, the highly flammable grass re-establishes rapidly and often expands, triggering a transition to a more frequent fire regime (Brooks et al., 2004; Evangelista et al., 2004; Floyd et al., 2006; Shinneman and Baker, 2009). The presence of cheatgrass would therefore support our short-term projection of increased fire risk, but would contradict our longer-term projection of reduced fire frequency.

4.2. Lower montane forests

Lower montane forests were historically situated at the peak of the curve in Fig. 2, and they were most frequently burned by surface fires with a correspondingly low mortality of dominant trees. However, in most areas, fire exclusion has resulted in dense forests that support greater incidence of stand-replacing fire (Arno et al., 1995; Allen et al., 2002; Agee and Skinner, 2005). We have high confidence that climate change will increase short-term fire risk regardless of stand structure because the existing fuels will be drier for more extended periods of time (Section 3). We also have high confidence that, on average, the short-term fire severity will increase because the drier fuels lead to more intense fires.

The longer-term trajectory for lower montane forests is less certain. One possible scenario is for fuel production, especially surface fuels, to be reduced. Historically, fires in the southern portion of the study area depended just as much on the fine fuel buildup during antecedent wet years as the occurrence of dry years (e.g. Brown and Wu, 2005). If such wet years become less frequent, there would be a corresponding reduction in the likelihood of broad scale burning (Swetnam and Betancourt, 1998; Donnegan et al., 2001; Westerling and Bryant, 2008; Littell et al., 2009). We project – with lower confidence – that lower montane forests will shift towards a lower fire frequency and a higher fire severity (Diggins et al., 2010). Uncertainty remains, however, about the balance of fuel condition and fuel production (the peak in Fig. 2) across the range of lower montane forests in the Rocky Mountains.

4.3. Upper montane forests

As in lower montane forests, management efforts have focused on restoring historical stand and landscape structures, but to date only a very small portion of affected areas have been treated (Schoennagel and Nelson, 2011). With high confidence we project that climate change will increase the short-term fire risk because fuels are plentiful and continuous and will be drier for longer periods of time during the longer fire season. From 1970 to 2005 the observed increase in large fire occurrence was most pronounced in the upper montane zone, particularly in the northern Rockies (Westerling et al., 2006). This increase was attributed to the increased fire season lengths, earlier snowmelt, and increased temperatures. We also have high confidence in the projection that fire severity will increase over the short term, with larger patches of high severity fire.

The longer-term trend in the fire regime of the upper montane is towards a more frequent fire regime, though this projection carries low confidence. These forests will still be relatively productive, with both high understory and overstory fuel loads. The longer-term projections about fire severity also carry low confidence, as they depend on the extent to which vegetation growth will be reduced due to more severe moisture deficits, or whether growth might increase as a result of a longer and warmer growing season. There is no consensus about the extent to which production in upper montane and subalpine forests is limited by moisture, and changes in tree growth with a future climate will likely vary by species (Kaufmann, 1985; Pataki et al., 2000). If fires indeed become more frequent, fire severity should decrease (e.g., Fig. 2).

However, if fuels dry more quickly without an associated decrease in fuel production, there would be an increase in severity along with an increase in fire frequency.

4.4. Subalpine forests

We are highly confident that the short-term impact of warming and drying will be an increase in fire risk, and a continuation of the high severity fire regime (Table 1). Over the longer-term, it is not clear how fuel production will change in response to the potentially conflicting increase in temperature, which should increase growth, and the greater potential for drying, which could decrease growth. The net balance will determine the resulting change in fuels. However, in contrast to the upper montane forest, the change in fuels is probably minor relative to the change in the amount and duration of drying, so we project with medium confidence that longer-term fire frequency will increase with the increase in temperature and shift from snow to rain. With medium confidence we also project that fire severity will remain high due to the hypothesized high vegetative productivity (Table 1).

Some have postulated a relatively large change in subalpine forests under a more frequent fire regime. Westerling et al. (2011) use downscaled global climate models and statistical relationships between climate conditions and past fires to predict a reduction in fire rotations from the historical value of more than 120 years to less than 20 years by midcentury, and less than 10 years by the end of this century. This sharp increase in fire frequency may preclude conifer regeneration and cause a vegetation shift to a fuel-limited grassland or shrubland. This demonstrates the potentially complex feedbacks between climate, fire regimes, and forest composition, and the resulting uncertainty in predicting future fire severity and frequency.

5. Fire management alternatives

In this section we briefly describe four alternatives for managing fire – fire suppression, wildfire with little to no active management, prescribed fire, and mechanical treatments – and the potential of these to mitigate climate change impacts to fire regimes and forests. The following sections then discuss the costs and benefits of these alternatives for four ecosystem services: carbon sequestration, water resources, air quality and human health, and biodiversity.

In the Rocky Mountain region fire suppression has been the dominant paradigm for fire management. Future fire management is expected to be similar given the resources at risk (Stephens et al., 2013). Suppression generally has been successful except during the most extreme weather conditions, but suppression is becoming less effective as drier conditions increase the frequency, extent, and severity of fires, and overwhelm both firefighting resources and the ability to safely put out fires (Adams, 2013).

Wildfires are fires that burn without effective fire suppression. Scientists, resource managers, and policy makers have increasingly been recognizing the benefits of letting some naturally-ignited fires, or parts of fires, burn when fewer resources are at risk and/or there may be some ecological benefits (Fire Executive Council, 2009). Prescribed fire is defined as the application of management-ignited fire to achieve pre-specified objectives, such as a reduction in surface and/or ladder fuels, restoration and maintenance of a historical fire regime, controlling weeds, or creating habitat. The widespread application of prescribed fire is increasingly limited by the risk of escape, agency budgets and staffing, the shortage of weather conditions suitable to achieving desired fire behavior, and the effects of smoke on public health and visibility (reviewed in Quinn-Davidson and Varner, 2012; Ryan et al., 2013). Although we do not know of any studies on how climate change will affect burn windows, we would project that there may not be much short-term change in the number of burn windows, but these will be shifted temporally (e.g., later in the fall, earlier in the spring).

Mechanical treatments typically thin canopy fuels to reduce the risk of crown fire, but sometimes they also are designed to reduce surface and/or ladder fuels (Agee and Skinner, 2005; Reinhardt et al., 2008; Stephens et al., 2012). Such treatments can mimic some of the beneficial effects of prescribed fire while avoiding most of the potential adverse effects. Mechanical treatments are primarily limited by funding rather than weather or air quality concerns (Hjerpe et al., 2009). While the cost of mechanical treatments is highly variable (Reinhardt et al., 2008), they often cost several times as much as prescribed fire (Calkin and Gebert, 2006). Timber sales, stewardship contracts, and biomass utilization are increasingly used to offset treatment costs (Nielsen-Pincus et al., 2013). Costs also can be reduced by shredding or chipping the material and leaving it on site rather than having to remove and dispose of the unmerchantable woody material (Wolk and Rocca, 2009; Battaglia et al., 2010). Piling and burning is another common approach to the disposal of unwanted biomass.

5.1. Potential for management to mitigate changes in fire regimes and vegetation

The suitability and limitations of fire suppression, wildfire, prescribed fire, and mechanical thinning treatments for mitigating changes in fire regimes and vegetation varies by forest type (Table 2) as described below. Fire suppression in persistent piñon-juniper woodlands generally has not caused much structural change because extensive, crown-replacing fires were historically infrequent (Baker and Shinneman, 2004; Floyd et al., 2004). Once burned, however, forest recovery may take decades to centuries because the dominant tree species are not adapted to regenerate after fire. The danger of the wildfire option is that future wildfires may be larger and more regionally synchronous. A warmer and drier climate is likely to further retard post-fire tree re-establishment, and increase the likelihood of a permanent type conversion to shrubland or grassland (Romme et al., 2003a). Prescribed surface fires and mechanical thinning are not attractive options for piñon-juniper woodlands since piñon and juniper do not tolerate fire well, and widely spaced trees were not the historical norm, at least in the majority of piñon-juniper ecosystems in the region (Floyd et al., 2004; Romme et al., 2009). Hence fire suppression, if effective, may be the best management strategy for maintaining piñon-juniper landscapes. The chances of effective fire suppression may be increased by creating fire breaks through strategically-located thinning treatments, but these would result in a novel community structure rather than recreating a historical one.

In contrast, fire exclusion has altered stand structure and forest heterogeneity in lower and upper montane forests (Arno et al., 1995; Brown et al., 1999; Kaufmann et al., 2000; Veblen et al., 2000; Allen et al., 2002). The historic policy of fire suppression has led to high fuel loadings in many areas, making fire suppression an increasingly untenable option, particularly under extreme weather conditions. Allowing wildfires to burn will often lead to undesirable levels of tree mortality unless weather conditions are mild or historical forest structure has been maintained or restored. The projected increase in stand-replacing fires also will alter forest structure and composition by selectively eliminating those tree species that are not adapted to regenerate after stand-replacing fires (Schoennagel et al., 2004). Restoration treatments that use prescribed fire or mechanical thinning to restore historical forest structure and landscape heterogeneity have the potential to reduce the chances of uncharacteristically severe and damaging wildfires

Table 2
Hypothesized effectiveness of land management alternatives to mitigate changes in fire regime and vegetation. *High* means the management practice will help maintain the ecosystem, *moderate* indicates a modest chance that the management practice could help maintain the ecosystem, *low* means the management practice is unlikely to help maintain the ecosystem, and *negative* means the management practice is likely to further exacerbate changes in fire regimes and/or vegetation.

Forest type	Fire suppression	Wildfire/No management intervention	Prescribed fire	Mechanical thinning
Piñon-juniper	Moderate	Negative	Low	Low/moderate
Lower montane	Negative	Negative (high where historical stand structure intact or restored)	High	High
Upper montane	Negative	Low (high where historical landscape structure intact or restored)	Moderate	Moderate
Subalpine	Low	High	Moderate	Low

(Finney et al., 2005; Stephens et al., 2012). The restoration of a frequent surface fire regime, with a component of crown fire in the upper montane, should help reduce the frequency and size of the more severe fires (Littell et al., 2009) that are projected to occur as a result of a warmer and drier climate.

The use of prescribed fire and mechanical treatments in the upper montane forests are projected to be moderately effective in mitigating potential climate change impacts by reducing the fire extent and severity (Table 2). Because fire regimes are somewhat moisture limited in the upper montane, thinning may be less effective than in the lower montane forests because this would subject the relatively abundant surface fuels to more wind and solar radiation, which could accelerate fuel drying and intensify fire behavior. Mechanical treatments that increase surface fuel loads are of particular concern (Graham et al., 2011). Alternatively, a more frequent fire regime in the upper montane may sufficiently reduce the surface and ladder fuels to shift the fire regime away from infrequent, stand-replacing fires to a fire regime that is more similar to the lower montane forests. Hence the most likely scenario of prescribed fire and mechanical treatments is to effectively prepare these forests for the coming changes in fire regimes and help maintain the ecosystem over the longer-term.

Fire suppression in subalpine forests is not detrimental to individual forest stands or landscape-scale forest composition in the short-term, given the historically long fire recurrence interval (Table 1). However, fire suppression over a longer time period would reduce the larger-scale heterogeneity in stand ages and species composition. Relative to the other forest types, and at least in the short-term, subalpine forests would benefit the most from allowing wildfires to burn, because this would increase landscape heterogeneity, benefitting species such as aspen and lodgepole pine that respond well to stand-replacing wildfire (Schoennagel et al., 2004; Sibold et al., 2006).

The use of prescribed fires in the subalpine is more difficult given the high fuel loadings, the uncertain acceptance by the public of initiating the stand-replacing fires that are characteristic of this forest type, and the associated effects on air and water quality as described in the follow sections. Management options are further limited because mechanical treatments do not have a natural analog in subalpine forests unless they involve clearcutting. Similar to the upper montane forests, an effort to create a more open canopy structure may exacerbate the effects of a warmer and drier climate by allowing an even greater drying of forest fuels with the associated effects on fire frequency and severity (Schoennagel et al., 2004). Hence the selection of management options is more difficult in the subalpine zone because none of the options can effectively help restore or mimic the natural fire regime without having other adverse effects.

5.2. Carbon sequestration

Wildland fires in the US have the potential to release large amounts of carbon to the atmosphere through the combustion of wildland fuels and subsequent decomposition of dead woody biomass. CO_2 emissions from vegetation fires in the five state region averaged 24.3 $TgCO_2\,yr^{-1}$ between 2001 and 2008 (Wiedinmyer and Hurteau, 2010) and play an increasingly significant role in the regional carbon budget; during the severe 2002 fire season in Colorado, C emissions from wildfire were comparable to those from the Colorado transportation sector (Brown, 2011). As vegetation regrows, however, forests typically recover the carbon that was emitted during the fire. For example, a Sierran mixed-conifer forest recovered its carbon in less than seven years after a low severity surface fire (Hurteau and North, 2010), a time span comparable to the historic fire return interval in such forests (Caprio and Swetnam, 1995). A lodgepole pine forests in Yellowstone National Park recovered about 90% of pre-fire carbon within 100 years following a stand-replacing fire, with a historic mean fire interval of 150–300 years (Kashian et al., 2013). Over time scales that encompass the entire cycle of fire to the regrowth of a mature forest, therefore, wildfires and prescribed fires should not lead to a net loss of carbon to the atmosphere (Campbell et al., 2012). However, it follows that if forests fail to recover following fire, if fire frequency is higher than the time it takes to recover carbon stocks, or if there is a permanent change in forest structure to a state with lower carbon stocks, there will be a net loss of carbon over time.

Failure to recover pre-fire carbon stocks might occur as a consequence of uncharacteristic fire regimes to which dominant plant species are not adapted. Fires that are too severe for mature adults or their propagules to survive, or fire return intervals that are out of sync with species' life histories, may trigger a type conversion to grassland or shrubland (Keeley and Brennan, 2012; Roccaforte et al., 2012). Alternately, an unfavorable post-fire environment due to decreased soil fertility or altered climate regimes could prevent pre-fire communities from recovering after fire (Romme et al., 2003a; Bormann et al., 2008).

Management activities that help to prevent such a type conversion may be appropriate options for preserving forest carbon storage capacity (Ryan et al., 2010; Campbell et al., 2012). Mechanical thinning and prescribed fire are touted as treatments that can reduce the risk of severe crown fire in montane forests (Hurteau and North, 2009; Wiedinmyer and Hurteau, 2010; Hurteau and Brooks, 2011). However, these treatments themselves remove biomass from the forest and emit C to the atmosphere. Prescribed fire releases C to the atmosphere through combustion, and thinning treatments reduce net C storage because most thinned biomass is either left on site to decompose or piled and burned (Ryan et al., 2010; Campbell et al., 2012). While forest C estimates in relation to fire and forest thinning are largely lacking for the Rocky Mountain region, a review of fuels treatment in forests similar to this region's lower montane forests shows aboveground C losses approximately 12% for prescribed fire and 30% for thinning (Campbell et al., 2012). Assuming such treatments are maintained in perpetuity, these losses in C storage will be sustained. Campbell et al. (2012) show that, as long as forests are expected to regrow following wildfire, the C cost of these treatments is not justified by the associated reduction in C emissions from wildfire.

Forest types that are considered to be resilient, or most likely to recover after a future fire, are assessed as having a low risk of carbon loss (Table 3) over the long term. Most subalpine forests fall into this category because the life history strategies of the dominant trees (i.e., serotinous cones in lodgepole pine, wind-dispersed seed in spruce, resprouting in aspen) allow them to regenerate quickly following stand-replacing wildfire. Two caveats are worthy of mention, however. First, several more narrowly distributed subalpine species are probably not resilient to wildfire, especially the five needle pines (whitebark pine, limber pine, western white pine, bristlecone pine). Second, it is possible that future fires will become so frequent that subalpine trees cannot complete a life cycle between fire intervals, as Westerling et al. (2011) predicted for Yellowstone subalpine forests. Even if subalpine forests were indeed vulnerable to type conversion in the next several decades, prescribed fire and mechanical thinning would be unlikely to mitigate this conversion because fire regimes in these forests are climate limited, not fuel limited (Fig. 2).

Forest types with a high probability of permanently changing state to a different vegetation type that stores less carbon include piñon-juniper woodlands and lower montane forests. In piñon-juniper woodlands, prescribed fire and thinning are unlikely to substantially alter the risk of severe fire (Tables 2 and 3), in part because any spreading fire, regardless of fuel load, would likely kill the trees and result in a slow biomass recovery. Lower montane forests, in contrast, would benefit from prescribed fire or thinning treatments that restore historical forest structure and increase the chance that mature trees and their seeds will survive wildfire. Upper montane forests fall between the lower montane and the subalpine in terms of their risk of permanently losing forest carbon stocks, and would likely benefit from landscape-scale restoration treatments that reduce the risk of extensive stand-replacing fires.

5.3. Water quantity and quality

The effects of wildfires on runoff and erosion rates is a major societal concern because of the potential for severe impacts on site productivity, peak flows, water quality, aquatic habitat, and reservoir sedimentation (e.g.; Moody and Martin, 2001; Shakesby and Doerr, 2006; Rhoades et al., 2011; Smith et al., 2011). From a management perspective, it also is important to compare the effect of wildfires on runoff and erosion relative to efforts to reduce fire risk, such as mechanical thinning and prescribed fire. Hence this section first discusses how forest fires alter runoff and erosion rates. Then it summarizes the respective effects of climate change, fire suppression, and heavy thinning on fire severity and frequency, and describes how these changes in turn alter runoff and erosion rates by forest type. In this section, fire severity is related to the magnitude of the changes in soil characteristics and ground cover (Parsons et al., 2010), as these are dominant controls on post-fire runoff and erosion rates (Benavides-Solorio and MacDonald, 2005; Larsen et al., 2009).

5.3.1. Effects of fires on runoff, erosion, and water quality

In most densely forested areas there is very little or no bare soil and nearly all of the rainfall and snowmelt infiltrates, so the primary runoff process is subsurface stormflow (e.g., Dunne and Leopold, 1978). This explains why forests are widely recognized for producing clean water while minimizing floods (National Research Council, 2008). High-severity forest fires are a great concern because they remove most of the protective ground cover and alter the surface soils (Neary et al., 2005; Larsen et al., 2009). This causes a change from subsurface flow to overland flow, and in the Colorado Front Range high-severity fires reduce infiltration rates from greater than 50 mm h^{-1} to only about 8–10 mm h^{-1} (Moody and Martin, 2001; Libohova, 2004; Neary et al., 2005; Kunze and Stednick, 2006). This decrease in infiltration is due to a series of often interacting processes, including: (1) burning the surface organic matter and the associated reduction in aggregate stability and increase in soil erodibility (Blake et al., 2007); (2) surface crusting as a result of raindrop impact on bare mineral soil (Larsen et al., 2009); (3) releasing various organic compounds that condense on the underlying soil particles to create a water repellent (hydrophobic) layer (Robichaud, 2000); and (4) reduction in surface roughness and the resultant increase in overland flow velocities (Lavee et al., 1995). The greater volume and speed of overland flow can cause extensive rilling, gullying and debris flows in steeper areas, and much of the coarse mineral sediment is deposited in lower-gradient downstream areas (Moody and Martin, 2001). In contrast, the ash and finer particles tend to remain in suspension, causing a degradation of water quality and greatly increasing water treatment costs (Smith et al., 2011). Wildfires can affect a series of other water quality parameters, including an increase in nitrates, phosphorus, and heavy metals, but these generally are of much less concern than the increases in ash and sediment concentrations (Neary et al., 2005; National Research Council, 2008; Smith et al., 2011).

Low-severity fires generally have a much smaller effect on runoff and erosion rates because these typically remove very little of the forest canopy and result in less than 30–40% bare soil (e.g.,Benavides-Solorio and MacDonald, 2005). Similarly, prescribed fires are typically designed to burn at low severity, with perhaps small patches of moderate severity, so prescribed fires should have minimal effect on runoff and erosion rates at anything larger than the hillslope scale. Runoff and erosion rates after moderate severity fires are harder to predict because they fall into the middle ground where the changes in runoff and erosion can range

Table 3
Risks of long-term loss of carbon stocks under different management alternatives. Generally, forests that grow back to pre-fire levels of biomass are considered not to be at risk for suffering significant loss in carbon stock. Forests at high risk of losing carbon stock are those that have high potential to undergo a type conversion after fire, to a vegetation type that supports lower biomass/less carbon.

	Fire suppression	Wildfire	Prescribed fire	Mechanical thinning
Piñon-juniper	If effective, retains largest carbon stock	High risk of loss	Carbon cost, unlikely to lower risk of large loss	Carbon cost, unlikely to lower risk of large loss
Lower montane	If effective, retains largest carbon stock	High risk of loss	Carbon cost, but may lower risk of large loss	Carbon cost, but may lower risk of large loss
Upper montane	If effective, retains largest carbon stock	Medium risk of loss	Carbon cost, but may lower risk of large loss	Carbon cost, but may lower risk of large loss
Subalpine	If effective, retains largest carbon stock	Low risk of loss	Low risk of loss	Low risk of loss

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from relatively small to quite severe, depending in large part on the amount and intensity of rainstorms in the first year or two after burning (Benavides-Solorio and MacDonald, 2005).

Wildfires in the lower montane forest are generally of greatest concern because these forests have sufficient fuel loadings to burn at high severity, tend to burn more frequently, and they are often adjacent to highly-populated areas. Annual rainfall and rainfall erosivity – which is a function of rainfall intensity and amount (Renard et al., 1997) – also tend to be higher in the lower montane forest than in either the drier piñon-juniper or the higher-elevation but snow-dominated forest types. More research has been done on post-fire runoff and erosion rates in the lower montane forests than other forest types; this research indicates that high-severity wildfires can increase the size of peak flows by a factor of 10 or more, and erosion rates can increase from near zero to more than 10 metric tons (Mg, or megagrams) per hectare per year (for reference, soil formation rates are closer to 0.1 Mg ha^{-1} yr^{-1}) (MacDonald and Stednick, 2003; Neary et al., 2005; Moody and Martin, 2009). Post-fire runoff and erosion rates tend to be greater in the central and southern Rocky Mountains where high-intensity summer convective storms are more common, and lower in the northern Rocky Mountains where there are fewer summer convective storms (Miller et al., 2011).

5.3.2. Climate change effects on the fire regime, runoff, and erosion

The projected changes in temperature and precipitation are likely to increase runoff and erosion rates because of the shift from snow to rain and greater storm intensities. Snowmelt rates are generally too low to cause much surface runoff and post-fire erosion at the hillslope scale (Benavides-Solorio and MacDonald, 2005), although the accumulated water can cause sediment transport and channel incision at larger scale. Of primary concern is the shift from snow to rain, as the raindrop impacts can detach soil particles and rainfall intensities can exceed the 8–10 mm h^{-1} needed to initiate overland flow.

For the piñon-juniper and lower montane forests climate change is projected to cause both an initial increase in soil burn severity and, due to warmer air temperatures, an increase in rainfall intensity. This combination will cause a short-term increase post-fire runoff and erosion rates, with a greater effect on erosion because of the large, non-linear increase in erosion with increasing rainfall intensities (Table 4). The projected effect is smaller in piñon-juniper than the lower montane forests because the lower productivity and associated lack of a deep, continuous litter layer will limit the effects of burning on soil organic matter, aggregate stability, and infiltration rates.

The effect of subsequent fires will vary according to the projected longer-term changes in forest productivity; in drier, more marginal areas the projected decrease in forest productivity and fuel loadings may decrease fire effects on soil properties and thereby decrease post-fire runoff and erosion rates. In most areas, particularly the northern Rockies, the warmer and slightly wetter conditions should increase forest productivity and the impact of fires on soils. This increase in severity, when combined with the projected increase in rainfall intensity, should increase both the size of peak flows and post-fire erosion rates, with a larger projected increase in erosion than runoff (Table 4).

For the upper montane forests, a warmer climate should increase the frequency of wildfires and slightly increase fire effects on soils. This increase in fire frequency, when combined with the expected increase in the amount and intensity of rain as well as the decrease in snow cover, may slightly increase annual water yields and cause a greater increase in the size of peak flows (Table 4). More importantly, we project that these changes will result in a much larger increase in post-fire erosion rates because post-fire erosion is so sensitive to fire severity and rainfall intensities (Benavides-Solorio and MacDonald, 2005; Wagenbrenner et al., submitted for publication) (Table 4).

Similarly, climate change in the sub-alpine zone is projected to generally increase forest productivity, fire frequency, and possibly fire severity. Again this increase in productivity plus the changes in climate should result in larger and more frequent post-fire peak flows and substantially higher erosion rates (Table 4). In both the upper montane and the sub-alpine zone the post-fire increases in runoff and erosion are due more to the increased amount and intensity of rainfall rather than the changes in primary productivity or fire severity (Miller et al., 2011). The relative importance of the projected increase in fire frequency is more difficult to quantify, but at least in the near term the increase in fire frequency will not be as important as the changes in the amount and intensity of rainfall.

5.3.3. Effects of fire suppression

The projected effects of fire suppression largely depend on how suppression alters fire frequency and severity over both the short- and long-term. A shift in the fire regime from frequent, low-to-moderate severity fires to less frequent, high-severity fires should increase runoff and erosion rates because the increase in fire severity has a greater effect on runoff and erosion rates than an increase in the frequency of lower severity fires. Hence fire suppression, when combined with climate change, is projected to increase long-term average erosion rates in the piñon-juniper and lower montane zones by increasing fire severity (Table 4). However, the effects of fire suppression are projected to be much smaller for piñon-juniper because forest productivity is too water-limited to support a major change in fire severity (Table 4). Any increase in forest density due to fire suppression is projected to have little or no effect on runoff in these two forest types because the piñon-juniper and lower montane forests already yield very little runoff, so a denser forest will not substantially alter the basic water balance (Table 4, MacDonald and Stednick, 2003).

In the upper montane and sub-alpine zones fire suppression should not have any effect on runoff because these forests already are sufficiently dense to maximize interception and evapotranspiration (MacDonald and Stednick, 2003). Fire suppression might cause a small increase in erosion if this increases fire severity,

Table 4
Projected changes in runoff, particularly peak flows, and erosion rates due to fire suppression, wildfires assuming no management actions, prescribed fires, and heavy mechanical thinning by forest type. *Q* refers to the amount of runoff, + indicates an increase, − indicates a decrease, and 0 indicates no change.

Forest type	Fire suppression		Wildfire		Prescribed fire		Mechanical thinning	
	ΔQ	ΔErosion	ΔQ	ΔErosion	ΔQ	ΔErosion	ΔQ	ΔErosion
Pinyon-juniper	0	Small +	0	Small -	0	Small +	0	0
Lower montane	0 or small −	+	0 or small +	Variable, but may +	0 or small +	0 or small +	0 or small +	0 or small +
Upper montane	0	Small +	+	++	0 or small +	0 or small +	0 or small +	0 or small +
Sub-alpine	0	Small +	+	++	0 or small +	0 or small +	0 or small +	0 or small +

but most wildfires in these zones are already high severity. Conversely, suppression could decrease the average long-term erosion rate if suppression can significantly reduce the frequency or extent of high severity fires without increasing fire severity.

5.3.4. Effects of thinning on runoff and erosion

There is an extensive literature on how forest harvest alters hydrologic and erosion processes, and this understanding can be used to project the likely effects of mechanical thinning treatments by forest type (National Research Council, 2008). The fundamental hydrologic principle is that removing part or all of the forest canopy will reduce interception and transpiration, and this can result in a detectable increase in water yields when annual precipitation is at least 450–500 mm (e.g., Bosch and Hewlett, 1982; National Research Council, 2008). Little or no change in annual water yields or peak flows can be expected if less than 15–20% of the vegetation canopy is removed, but this threshold may be higher in drier areas (e.g., annual precipitation less than 500–1000 mm yr^{-1}) because the remaining vegetation will use much of the "saved" water (MacDonald and Stednick, 2003). This means that thinning, if done with minimal soil disturbance, will have virtually no effect on the amount or timing of runoff in the piñon-juniper zone, and little to no effect in the lower montane zone (Table 4, Troendle et al., 2010). Heavy thinning, such as selectively removing 25–40% of the forest canopy, may slightly increase water yields and snowmelt peak flows in the upper montane and sub-alpine zones (MacDonald and Stednick, 2003; Troendle et al., 2010), but this increase generally will be small compared to the inter-annual variability in runoff and rapidly eliminated with vegetative regrowth and opportunistic water uptake (Table 4).

Similarly, mechanical treatments generally should not increase erosion rates (Robichaud et al., 2010). More specifically, as long as the treatments do not compact the soil, cause more than 30–35% bare soil, create an extensive road or skid trail network, or disturb ephemeral channels or streambanks, there should not be any detectable increase in erosion or degradation of water quality (e.g.; Cram et al., 2007; Karwan et al., 2007; Table 4). Alternatively, the extensive use of heavy machinery, particularly on moist soils, can compact the soils to the extent that snowmelt or rainstorms can induce overland flow with a resultant increase in surface erosion. Hence the effects of thinning on runoff and erosion are highly dependent on the site-scale conditions and practices, but typically become insignificant at larger scales and in relation to the effects of moderate or high severity wildfires (Robichaud et al., 2010; Wagenbrenner et al., submitted for publication).

5.4. Air quality and human health

Smoke generated from wildland fires is an emerging concern because of its impacts on atmospheric visibility, human health, and air quality regulatory compliance, and climate feedbacks (particularly via cloud impacts) (Wotawa and Trainer, 2000; Wu et al., 2008; Oltmans et al., 2010; O'Neill et al., 2013). Smoke composition depends upon both fuels and combustion conditions, as discussed below, but biomass smoke contains a large number of gas-phase and particulate species (Reid et al., 2005; McMeeking et al., 2009). The importance of air quality considerations varies greatly with proximity to population centers and local weather patterns.

Epidemiological studies and hospital admission studies indicate that biomass smoke is associated with human health impacts (Duclos et al., 1990; Naeher et al., 2007). Two important criteria air pollutants impacted by smoke include particulate matter with diameters less than 2.5 micrometers (PM2.5) and tropospheric ozone (O_3) via emission of precursors (Jaffe et al., 2008a; Jaffe et al., 2008b). Biomass burning emissions of hydrocarbons and NOx are a large contributor to the formation of O_3 and exceedances of the National Ambient Air Quality Standards (NAAQS) for O_3 (Oltmans et al., 2010).

Atmospheric visibility over large regions can be severely affected by wildfire emissions (McMeeking et al., 2006). Spracklen et al. (2007, 2009) concluded that summer wildfires were the most important driver of inter-annual variability in observed total carbonaceous PM across the continental U.S. These are largely due to increased area burned, but also because of altered weather patterns (Spracklen et al., 2009). Similarly, a modeling study by Park et al. (2007) attributed 50% of U.S. annual mean particulate carbon concentrations to biomass burning. Extreme visibility impairment can jeopardize transportation and traffic safety (Berbery et al., 2008).

The severity of air quality impacts is controlled to a large extent by the microphysical and chemical properties of smoke, such as particle sizing (McMeeking et al., 2009; Levin et al., 2010). The effects of smoke on air quality also depend on meteorological parameters such as relative humidity, which impacts particle size (Carrico et al., 2005; Carrico et al., 2010), and the optical properties of the smoke particles (Lewis et al., 2009). Particle size and structure plays an important role in the cloud nucleating properties of smoke for both freezing and non-freezing clouds (DeMott et al., 2009; Petters et al., 2009). The size distribution of particles is also intimately connected to human exposure endpoints, and ultrafine particles (diameter $Dp < 100$ nm) have been gaining increased attention due to their adverse effects on health (Pope and Dockery, 2006; Pope et al., 2009).

Air quality impacts are directly related to fire size and intensity, as well as the fuels being consumed, but these impacts are still difficult to predict. A laboratory study of 255 controlled burns with 33 plant species showed that mass emission factors of PM2.5 varied from 2 to 82 g of PM2.5 per kg of dry fuel; long-needled conifers had the highest group average at 29 ± 25 g kg^{-1}) (McMeeking et al., 2009). However, the combustion efficiency and fire phase was more important than the fuel type. The flaming vs. smoldering behavior of a fire can be quantified by the Modified Combustion Efficiency ($MCE = \Delta CO_2/(\Delta CO_2 + \Delta CO)$); MCE values greater than 0.9 are considered predominantly flaming (Reid et al., 2005). PM2.5 emissions strongly increases as MCE decreases, indicating the greater potential for adverse air quality impacts from smoldering fires (McMeeking et al., 2009). Fuel moisture content is an important control on the MCE and hence PM 2.5 emissions (Chen et al., 2010). The aerosol single scattering albedo, which is important to aerosol climate forcing, also varies with MCE, as it is close to 1 (cooling effect) for smoldering combustion and less than 0.5 (warming effect) for flaming combustion (McMeeking et al., 2009).

Freshly emitted smoke aerosols undergo physical transformations and chemical aging as they are transported from source regions to downwind impacted regions (e.g. secondary organic aerosol formation and oxidation) (Hennigan et al., 2011; Pratt et al., 2011; Martin et al., 2013). Two further factors may come into play and as the smoke aerosol ages: a decrease in MCE is linked to a higher NH_3 to total N emission ratio, and larger emission factors for most gas-phase hydrocarbon species (McMeeking et al., 2009). Both favor secondary PM formation via formation of ammoniated salt species and secondary organic aerosols, respectively.

Management alternatives vary in terms of their consequences for air quality and human health. Mechanical treatments and effective fire suppression have little or no impacts on air quality, while wildfires clearly have severe impacts on air quality, with little warning to mitigate impacts on sensitive populations. Hence any reduction in fire frequency, severity, and extent due to mechanical thinning and, to a lesser extent, prescribed fire can reduce air quality impacts. In lower montane and upper montane forests, restoration of a historical forest structure and the fire process by these treatments is likely to decrease the risk of severe and extensive wildfires, and hence the air quality impacts. In piñon-juniper and subalpine forests, mechanical thinning and prescribed fire will have a smaller effect on future wildfires and hence have less of an air quality benefit (Table 2). In lower and upper montane forests prescribed fire should substitute limited and somewhat controllable air quality impacts over repeated burn periods for severe, unplanned air quality events. Smoke concerns from prescribed fires can usually be minimized, but not eliminated, by burning when atmospheric conditions are more conducive to smoke dispersion. The fuel and atmospheric conditions largely control whether a fire is flaming or smoldering and, again, this can be partially controlled by the choice of conditions for burning. The problem is that the conditions that may be most conducive to reducing air quality impacts (high dispersal and flaming rather than smoldering combustion) may not always be consistent with prescribed fire objectives, especially reducing the risk of fire escape.

5.5. Conservation of biodiversity

Plant and animal species that depend on forested ecosystems will have to respond to both the direct impacts of the future climate and to the indirect climate impacts brought by habitat changes and altered fire regimes. The response of individual species will often be difficult to predict, and management alternatives will likely need to be assessed on a species- and location-specific basis (McKenzie et al., 2004). It is reasonable to assume, however, that restoration and maintenance of a fire regime within the historical range of variability of a forested landscape would be the best way to mimic the conditions that supported diverse suites of flora and fauna during their evolutionary histories. Further, maintaining high spatial variability within the historical range of variability is essential to preserving the habitat requirements of a diverse range of plant and animal species (Allen et al., 2002; Kotliar et al., 2007; Floyd, 2003). The comparison of management alternatives in Table 2, therefore, largely summarizes the tradeoffs that would be expected between approaches in terms of biodiversity. Research on the effects of climate change and fire management alternatives on biodiversity in the Rocky Mountain region is scarce, but here we summarize the research results from recent studies.

5.5.1. Piñon-juniper woodlands

The complex structure of persistent piñon-juniper woodlands on the Colorado Plateau supports a high diversity of avian, mammal, and herbaceous species richness including several endemic or endangered plants (Floyd, 2003). Several plants that germinate following fire are also native to piñon-juniper woodlands in the region (Floyd and Colyer, 2003). Unfortunately, fires also allow for the invasion of exotic plant species such as thistles, knapweeds, and cheatgrass, which threaten native biodiversity (Romme et al., 2003b; Floyd et al., 2006). Drastic changes to fire regimes brought by climate change and/or cheatgrass invasion have the potential to threaten this native biodiversity. However, it is unlikely that management strategies other than fire suppression can prevent the extensive fires that may lead eventually to a loss of piñon-juniper woodland habitat. Further, prescribed fire and mechanical thinning may have unintended negative consequences. For example, prescribed fires may kill old piñon and juniper trees, which are not adapted to fire, changing the ecosystem from a woodland to a more open savanna and, eventually, a shrubland (Floyd, 2003), altering the habitat for many animal species. Cheatgrass invasion is also likely to be exacerbated by prescribed fire activities. Mechanical treatments are also often associated with cheatgrass invasion (Owen et al., 2009; Ross et al., 2012) and loss of habitat for woodland obligate animals (Crow and van Riper, 2010).

5.5.2. Lower, upper montane, and subalpine forests

Lower montane forests are believed to have supported a diverse understory flora, but studies of reference conditions for ponderosa forests come mostly from studies in the Southwest region. Nevertheless, studies of restoration treatments that use mechanical thinning and/or prescribed fire have shown an increase in understory diversity relative to untreated forests in the lower montane (Dodson et al., 2007; Wolk and Rocca, 2009). Findings from a recent experiment on the effects of prescribed fire and mechanical treatment showed that thinning and prescribed fire lead to different understory floras, and that mechanical treatment followed by prescribed fire led to the highest native species diversity, but also the most invasive species (Dodson and Fiedler, 2006; Metlen and Fiedler, 2006). Wildfire in the lower montane appears to favor exotic plant species, especially where fire severity is high (Hunter et al., 2006; Freeman et al., 2007; Fornwalt et al., 2010). There are very few studies on the effects of alternative management practices on herbaceous biodiversity in upper montane and subalpine forests in the Rocky Mountain region. However, wildfire has been shown to benefit plant diversity following subalpine forest fires in the greater Yellowstone area (e.g., Turner et al., 1997; Doyle et al., 1998).

Wildlife species including ungulates, small mammals, and birds respond strongly to the diversity of stand structures created by low-severity and high-severity fires across all Rocky Mountain forest types. Wildlife species including Canada lynx (*Lynx canadensis*; Ruggiero et al., 2000), elk (*Cervus elaphus*; Pearson et al., 1995), and American three-toed woodpecker (*Picoides dorsalis*; Kotliar et al., 2008) respond to spatial heterogeneity in fire severity and/or forest structure at a variety of spatial scales. Such complex patterns of fire severities are difficult to replicate in planned treatments (Kotliar et al., 2007). Efforts to introduce low severity surface fires in lower montane landscapes will not be sufficient to create the habitat required by bird species that associate with stand-replacement burns or the variety of wildlife species that require undisturbed forest habitats. Therefore, a diversity of management approaches, including high-severity wildfire, may be necessary to create habitat for the full suite of avian and mammalian biodiversity (Hutto, 1995; Kotliar et al., 2007; Zwolak and Foresman, 2007). The upper montane zone would be a prime area for implementing treatments designed to create landscape heterogeneity since a mixture of stand structures historically characterized the forests there.

6. Conclusions

The objectives of this paper were: (1) to characterize the likely short- and longer-term effects of projected climate changes on fuel dynamics and fire regimes for four generalized forest types in the Rocky Mountain region; (2) to review how these changes are likely to affect carbon sequestration, water resources, air quality, and biodiversity; and (3) to assess the suitability of four different management alternatives to mitigate these effects and maintain forest ecosystem services. By necessity we take a broad-based approach, and this also is appropriate given the variation in forest conditions and the fact that future climate projections show a clear warming but do not show a definitive trend in precipitation. The consensus of forecasts for a warmer climate is a short-term increase in fire risk for each of the four main forest types, with the largest increases in the southern part of the region. Climate change also is projected to cause a short-term increase fire severity in the lower and upper montane forests, exacerbated by changes in fuel loads and stand structures resulting from past fire exclusion. Fire severity will not greatly increase in piñon-juniper woodlands and subalpine forests as historical fires were already predominately high-severity.

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Over the longer term, after the next fire modifies existing fuel loads, the projections for future fire regimes are less certain due in large part to the uncertainty in the forecasts for future precipitation, the role of invasive annual grasses, future land management and land use changes, and how the changes in climate, fire frequency and severity will affect forest productivity. Hence, climate change may either enhance or mitigate fire risk, depending on whether fuels become dryer or wetter in each location and whether the increased warming causes a decrease in forest growth due to drying and increased fire frequencies, or an increase in growth due to warming, particularly at higher elevations. In fuel-limited ecosystems such as piñon-juniper and portions of the lower montane, fires are projected to become less frequent since these forests are likely to become less productive due to the warmer and possibly drier conditions. In forest types where fires are climate-limited, such as the upper montane and subalpine, fire frequency may increase as fuel production does not limit fire occurrence, and the fire season will increase in length and severity due to the lighter snowpack, earlier snowmelt, increased fire season length, and warmer temperatures. These changes will almost certainly overwhelm any possible increases in precipitation.

We also assessed how each these projected trajectories in fire frequency and severity are likely to affect four key ecosystem services: carbon sequestration, runoff and water quality, air quality, and biodiversity. To some extent, impacts on ecosystem services can be influenced by future forest management alternatives. In piñon-juniper, management actions that prevent wildfire and increase fire suppression effectiveness would help to preserve woodland structure and its associated biodiversity, limit opportunities for annual grass invasion, maintain carbon stocks, and preserve water and air quality. In lower and upper montane forests, both prescribed fire and mechanical treatments have the potential to restore and maintain historical forest structure, promote native diversity, and lower the risk of severe air quality events. Treated forests will store less carbon than untreated forests but, in montane ecosystems, should be more resistant to type conversion and the associated long-term loss of carbon stock. Where feasible, prescribed fire in these forests may be the better choice for mimicking a natural process and promoting biodiversity, whereas mechanical treatments produce many of the benefits of prescribed fire without associated air quality concerns or the risk of fire escape. Finally, in subalpine forests, allowing wildfires to burn where possible would mimic the natural fire process in these systems, benefit forest species, and maintain carbon stores. However, wildfires come with drawbacks in terms of potential increases in erosion and episodes of poor air quality.

The feasibility of alternative fire management approaches will be greatly constrained by local considerations such as land ownership patterns, agency budgets and logistics, federal and local policies, tolerance of risk, and landscape context. Further, our assessment shows that there tend to be tradeoffs among management alternatives, and that often no single management strategy will simultaneously optimize each of the four ecosystem services considered here. Nevertheless, some management approaches may be more beneficial than others for conserving forest ecosystem services, and these will vary widely by forest type. Key research needs include the relative roles of fuel versus climate limitation in the risk, occurrence, and severity of future fires across the forested zones of the Rocky Mountains, and how land management activities will interact with climate to impact forest ecosystem services in the region. Despite remaining uncertainties, our current knowledge is sufficient to make informed management decisions that reasonably weigh the likely consequences of alternative management strategies under a future climate.

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