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Soil greenhouse gas, carbon content, and tree growth response to biochar amendment in western United States forests

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Abstract

Restoring overstocked forests by thinning and pyrolyzing residual biomass produces biochar and other value-added products. Forest soils amended with biochar have potential to sequester carbon (C), improve soil quality, and alter greenhouse gas (GHG) emissions without depleting nutrient stocks. Yet, few studies have examined the effects of biochar on GHG emissions and tree growth in temperate forest soils. We measured GHG emissions, soil C content, and tree growth at managed forest sites in Idaho, Montana, and Oregon. We applied biochar amendments of 0, 2.5, or 25 Mg/ha to the forest soil surface. Flux of carbon dioxide and methane varied by season; however, neither were affected by biochar amendment. Flux of nitrous oxide was not detected at these nitrogen-limited and unfertilized forest sites. Biochar amendment increased soil C content by 41% but did not affect tree growth. Overall, biochar had no detrimental effects on forest trees or soils. We conclude that biochar can be used harmlessly for climate change mitigation in forests by sequestering C in the soil.

KEYWORDS

carbon dioxide, conifer, methane, overstocking, recalcitrant soil carbon, thinning

1 | INTRODUCTION

Many forests in the western United States are in need of restoration due to overstocking relating to lack of harvest and effective fire suppression (Weatherspoon & Skinner, 2002). Overstocking can increase the risk of wildfire (Schoennagel, Veblen, & Romme, 2004) because of smaller tree size and increased accumulaton of surface and ladder fuels (Lydersen, North, Knapp, & Collins, 2013), which can lead to intense wildfires. Forests can be thinned to reduce wildfire risks and to supply wood for bioenergy production (McIver et al., 2003). Thinning concentrates growth into merchantable trees by removing undesirable and suppressed trees. Thinning decreases risk of fire, insects, and drought while improving soil water and nutrient availability (Chase, Kimsey, Shaw, & Coleman, 2016; Ostaff et al., 2006; Zeide 2001). However, site nutrient capital will decrease if thinning residue is removed from the forest (Helmisaari et al., 2011; Jacobson, Kukkola, Malkonen, & Tveite, 2000) or piled (concentrated) and burned (Kalabokidis & Omi, 1998). Instead of burning slash piles, an alternative is to generate biochar from the waste wood (Page-Dumroese, Busse, Archuleta, McAvoy, & Roussel, 2017; Page-Dumroese, Jurgensen, & Terry, 2010) and return it to the forest. Biochar is charcoal created by pyrolysis (Bridgewater, 2004) and intentionally applied to improve inherent soil properties (Lehmann & Joseph, 2009), to restore soil functions, or increase ecosystem services (e.g., water filtration, carbon sequestration). Returning biochar made from forest residues to the surface of the forest floor or mineral soil returns nutrients that were removed during thinning because biochar contains most of the nutrients from the feedstock source (Gaskin, Steiner, Harris, Das, & Bibens, 2008).

Biochar made by fast pyrolysis, and added to forest soils can have similar properties as charcoal produced during a wildfire (DeLuca & Aplet, 2008; Harvey, Larsen, & Jurgensen, 1979; Matovic, 2011). Applying biochar can improve soil bulk density, porosity, moisture holding capacity, infiltration, and hydraulic conductivity (Atkinson, Fitzgerald, & Hipps, 2010; Ippolito, Laird, & Busscher, 2012; Mukherjee & Lal, 2013) and could improve tree seedling growth (Robertson, Rutherford, Lopez-Gutierrez, & Massicotte, 2012) and aid in forest restoration (Thomas & Gale, 2015).

Biochar mitigates climate change by sequestering carbon (C) (Lehmann, 2007) through its inherent resistance to microbial decomposition and long residence time. Biochar is resistant to microbial decomposition because of its condensed aromatic structure (Baldock & Smernik, 2002). This stability could further increase through interactions between biochar and soil minerals (Brodowski, John, Flessa, & Amelung, 2006). Naturally occurring charcoal from wildfires and human-created charcoal-rich soils (Anthrosols) can last for hundreds to thousands of years (Agee, 1996; Wang, Xiong, & Kuzyakov, 2016). Biochar's long residence time makes it a C sequestration and climate change mitigation tool (Lehmann, 2007; Wang et al., 2016).

Biochar has variable effects on greenhouse gases (GHGs), such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) flux when measured in agricultural soil, where biochar is mixed into the soil profile (Case, McNamara, Reay, & Whitaker, 2014; Spokas, Koskinen, Baker, & Reicosky, 2009; Spokas & Reicosky, 2009; van Zwieten et al., 2010). However, there is a lack of published research related to biochar applied to forest soil litter and related GHG flux changes, especially from field-based trials.

Biochar application to forest field-based trials is very different from agricultural field-based trials due to the nature of the biochar application. Application to agricultural soils involves mixing the biochar into the soil profile while in forest soils it is not possible to till the soil, so biochar is applied to the soil surface, where the biochar initially affects the forest soil and not the mineral soil. As the biochar makes its way from the surface to the mineral soil, GHG flux could change by impacting different soil processes. In addition, agricultural soils commonly have near neutral pH while forest soils are typically acidic. Biochar can affect soils differently because of the pH of soil relative that of biochar (Yuan & Xu, 2011). Thus far, field-based studies of biochar effects on GHG flux from forest soils have found no effect from biochar amendment in a northern temperate forest on CO_2, CH_4, and N_2O fluxes (Sackett et al., 2015) and no effect on CO_2 flux in a subtropical forest (Zhou et al., 2017). If biochar has little to no effect on GHG efflux, then it will be possible to conclude that biochar amendments to forests can effectively mitigate climate change if effects on forest productivity are neutral to positive.

Biochar has been shown to increase plant productivity in agriculture systems (Biederman & Harpole, 2013; Liu et al., 2013) possibly due to biochar improving soil water holding capacity (Laird, Fleming, Davis et al., 2010), soil liming (Biederman & Harpole, 2013), and decreased nutrient leaching (Laird, Fleming, Wang, Horton, & Karlen, 2010). Thomas and Gale (2015) performed a meta-analysis on tree growth responses to biochar, based primarily on seedling studies. It was concluded that there is a potential for large tree growth responses to biochar (a mean 41% increase in biomass), but growth rates were highly variable overall.

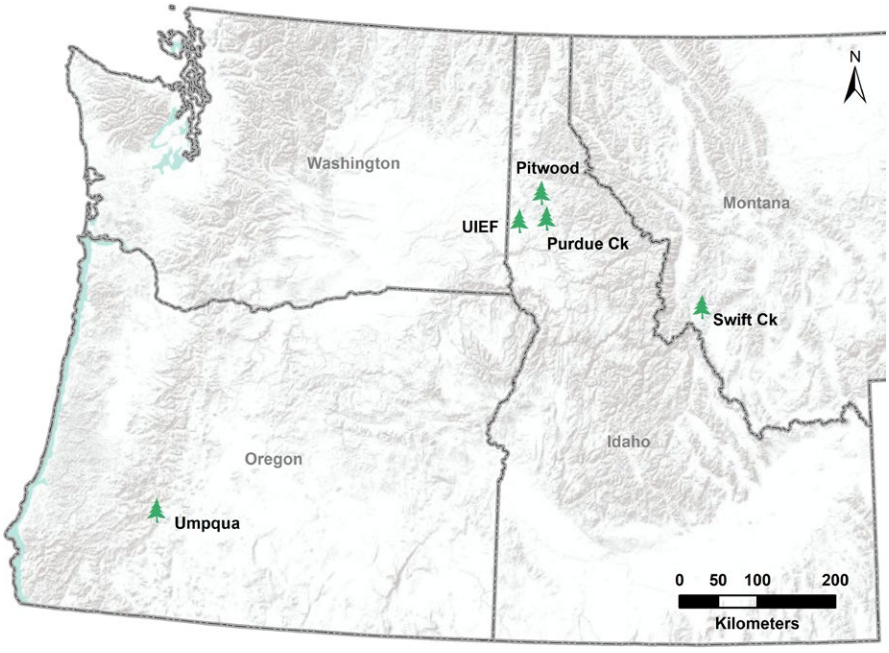

FIGURE 1 Biochar-amended forest sites in northwestern USA

TABLE 1 Physiographic, climatic, and experimental study site information

Site	Coordinates	Elevation (m)	Mean annual temperature[a] (°C)	Mean annual precipitation[a] (mm)	Installation year	Plot size[b] (m^2)	Plots per treatment	Biochar rate (Mg/ha)
Swift Creek	45.8928°*N* 113.7717°W	1,844	3.9	623	2010	57	6	0, 2.5, 25
UIEF	46.8495°*N* 116.8401°W	860	7.6	830	2014	1,600	8	0, 2.5
Purdue Creek	46.8858°*N* 116.3830°W	976	6.5	949	2011	49	5	0, 25
Pitwood	46.9831°*N* 116.4839°W	1,000	6.4	1,136	2014	1,600	8	0, 2.5
Umpqua	43.2347°*N* 122.3983°W	936	9.5	1,268	2009	29	5	0, 2.5, 25

[a]30-year average data from (PRISM Climate Group, 2018). [b]Plot size refers to the treated area. Umpqua, Swift Creek, and Purdue Creek were single tree plots meaning the biochar was applied to one tree, dependent on its crown radius. Pitwood and UIEF were 40 × 40 m plots containing several trees.

TABLE 2 Study site tree, stocking, and soil information

Site	Dominant tree species	Initial tree age (years)	Initial tree DBH[a] (cm)	TPH (trees/ha)	Soil texture	Soil porosity (%)	Soil organic matter (%)	Soil pH
Swift Creek	PP	44	27	175	Coarse loamy	49.8	8.4	6.2
UIEF	DF, GP, LPP, WL	27	12	373, 1,563	Silt loam	60.3	6.9	5.6
Purdue Creek	DF	22	13	203	Silt loam	71.2	9.4	6.4
Pitwood	DF, GF, WRC	20	14	467, 2,625	Silty clay loam	72.4	9.8	5.8
Umpqua	PP	13	10	342	Coarse	77.7	11.9	6.3

DF: Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Mirb.) Franco); GF: grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.); LPP: lodgepole pine (*Pinus contorta* Douglas ex Loudon); PP: ponderosa pine (*Pinus ponderosa* Douglas ex C. Lawson); WL: western larch (*Larix occidentalis* Nutt.); WRC: western red cedar (*Thuja plicata* Donn ex D. Don).

[a]DBH measurements are based on the average of 18 trees at Swift Creek, 171 trees at UIEF, 10 trees at Purdue Creek, 225 trees at Pitwood, and 15 trees at Umpqua.

TABLE 3 Physical and chemical characteristics of biochar used to amend forest sites in the western USA

Biochar Producer	pH	C (mg g^{-1})	N (mg g^{-1})	Ash (mg g^{-1})	Bulk density (g cm^{-3})	EC (mmhos cm^{-1})
Biochar Solutions	7.6	837	4.3	94	0.17	0.27
Evergreen Forest Products	8.3	257	0.8	403	0.17	0.29
Dynamotive CQuest	5.8	689	3.2	85	0.40	0.72

Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) seedling growth can decrease by 57%–69% depending on biochar application rate, (Sarauer & Coleman, 2018a). Seedling growth rates are higher in boreal forests and for angiosperms compared to temperate forests and conifers (Thomas & Gale, 2015), but there is high variability of plant growth responses to biochar applications in general (Spokas et al., 2012). There is a need for long-term field trials to investigate biochar's effects on forests because responses from short-term laboratory or greenhouse studies are not always comparable to field responses (Page-Dumroese, Coleman, & Thomas, 2016). Thus, the objective of this study was to examine the longer-term effects of biochar as a soil amendment on soil GHG emissions, soil C content, and tree growth in temperate, mixed-conifer forests in the western USA to test if biochar can be harmlessly used to mitigate climate change.

2 | MATERIALS AND METHODS

2.1 | Study sites

The study was conducted at five sites across the western USA (Figure 1, Tables 1 and 2). All study sites were part of previous or existing research and were established during thinning of field plots between 2009 and 2014 (Table 1). The Swift Creek site, located in Montana, was easternmost and at the highest elevation with low temperatures and precipitation. The Idaho sites (University of Idaho Experimental Forest [UIEF], Purdue Creek, and Pitwood) were intermediate compared to the Swift Creek and Umpqua locations. The Umpqua site, located in Oregon, was the southernmost location and had the warmest and wettest conditions. Soil varied from site to site (Table 2). Swift Creek soils were sandy skeletal, mixed, frigid Typic Haplustepts from the Totelake Series (Soil Survey Staff, 2012). UIEF soil was coarse-silty, mixed, superactive, frigid Vitrandic Fragixeralfs (Santa Series) (Soil Survey Staff, 2012) and more information about its study sites and treatments can be found in Sherman, Page-Dumroese, and Coleman (2018). The current study utilized the control and biochar only amended plots (across all thinning treatments) described in Sherman et al. (2018). Purdue Creek soils consisted of a Threebear-Norwidge Complex of medial over loamy, amorphic over mixed, superactive, frigid Oxyaquic Udivitrands (Threebear Series) and Alfic Udivitrands (Norwidge Series) (Soil Survey Staff, 2012). The Pitwood site was ashy over loamy, amorphic over isotic, frigid Typic Udivitrands (Flewsie Series) and more details about the Pitwood study sites and treatments can be found in Sherman et al. (2018). Umpqua soils were ashy-pumiceous, glassy Xeric Vitricryands from the Lapine Series (Soil Survey Staff, 2012).

2.2 | Biochar amendment

Biochar was applied uniformly to the soil surface at three rates: 0, 2.5, or 25 Mg/ha (Table 1) during site establishment. The low biochar amendment rate was intended to represent the amount produced from harvesting 10 Mg/ha dry biomass and 25% biochar conversion efficiency. The high rate was equivalent to those applied in agricultural settings. Biochar was hydraulically sprayed on to the soil surface at UIEF while it was applied manually at all other sites. Swift Creek received Biochar Solutions biochar (Carbondale, CO, USA) created via fast pyrolysis. The Idaho sites (UIEF, Purdue Creek, and Pitwood) received a high ash content biochar, Evergreen Forest Products biochar (New Meadows, ID, USA), which was created in a steam boiler (Table 3), and Umpqua received Dynamotive CQuest biochar (Richmond, BC, Canada) created via fast pyrolysis. We acknowledge that these differences in biochar properties may have variable impacts on tree growth and GHG flux.

2.3 | Soil carbon content

Soil samples were collected in each plot from 0 to 15 cm depth below the forest floor using a 5.5 cm diameter auger (AMS Inc, American Falls, ID, USA) several years after initial biochar application (exact number varies by site). Soils were dried at 60°C to a constant weight, sieved to 2 mm, and pulverized to a fine powder during 48 hr on an orbital shaker table (New Brunswick Scientific Co., New Brunswick, NJ, USA). Once samples were a fine powder, they were analyzed for C content with a Costech elemental analyzer (Costech Analytical, Valencia, CA, USA). Soil bulk density values were obtained from field collection for each plot using a bulk density sampler (AMS Inc). Bulk density was determined on the whole sample (with rocks) for a total bulk density. Soil C content was calculated by multiplying C concentration by fine fraction bulk density. To determine fine fraction bulk density, soils were sieved

at 2 mm and the remaining rock fragments were weighed. The rock fragment fraction and rock fragment density were determined, and then fine fraction bulk density was calculated based on the equations of Andraski (1991). To obtain soil C content on a land area basis, C content values were multiplied by 15 cm sampling depth and expressed on a square meter basis.

2.4 | Greenhouse gas measurements

Soil CO_2, CH_4, and N_2O fluxes were measured in spring (May), summer (July), and fall (October/November) 2015 at undisturbed field sites during the period of 10 a.m. to 3 p.m. Soil CO_2 efflux was collected using an LI-6400 with a 6400-09 soil chamber (LICOR, Lincoln, NE, USA). The soil chamber covers 71.6 cm^2 of soil and before insertion has a volume of 991 cm^3. The soil chamber was inserted one cm directly into the ground and the instrument took direct, real-time soil CO_2 efflux measurements.

Methane and N_2O fluxes were measured using a gradient method (Maier & Schack-Kirchner, 2014). Gradient profile sampling wells, sample analysis, and flux calculations have been described previously in Sarauer and Coleman (2018b) and an example of a gradient profile sampling well can be seen in Figure S1. Briefly, four gradient profile sampling wells were installed in each plot at 18 cm below the soil surface. Soil gas was collected from each sampling well and from the soil surface with a 25 ml gas-tight syringe (Hamilton Co., Reno, NV, USA). A 200 µl soil gas sample was manually injected into a gas chromatography mass spectrometer (Focus GC, ISQ MS, TG-Bond-Q-30C SIM column, Thermo Scientific, Waltham, MA, USA) to determine both CH_4 and N_2O gas concentration. The method of Tang, Baldocchi, Qi, and Xu (2003) was followed to calculate gas flux using Fick's first law of diffusion and Graham's Law of Diffusion, which requires diffusion coefficients. Specific equations can be found in Supporting Information Table S1. To obtain gas diffusion coefficients in the soil, measurements of soil temperature at 10 cm were obtained with an Omega Engineering thermocouple probe (Stamford, CT, USA) and atmospheric air pressure was measured with a Kestrel weather meter (Nielsen-Kellerman, Boothwyn, PA, USA). The gas tortuosity factor was computed using the Millington–Quirk model (Millington & Quirk, 1961) where the typical particle density of 2.65 g/cm^3 was assumed for mineral soils (Weil & Brady, 2017) and for andic soils (Bielders, Backer, & Delvaux, 1990; Maeda, Takenaka, & Warkentin, 1977). Volumetric water content was measured with a TRIME T3 soil access probe (Mesa Systems Co, Stonington, CT, USA) using access tubes installed in each plot. Bulk density was collected as described previously. Individual measurements from the gradient profiles were averaged together for each plot.

2.5 | Tree diameter growth

Tree diameter at breast height (DBH) was measured prior to biochar application and each fall thereafter. Each tree in the single tree plot locations was measured (Purdue Creek, Swift Creek, and Umpqua). All plot trees were measured at Pitwood and UIEF if they were over 2.5 cm in diameter, see Sherman et al. (2018) for more details. Diameter growth increment was the difference between initial and final DBH measurement divided by time (years).

2.6 | Statistical analysis

The effect of biochar treatment level, season, and their interactions on soil GHG fluxes, temperature, and moisture were tested in a two-way factorial analysis using Type III test of fixed effects to account for unbalanced design. A repeated measures mixed model, with a compound symmetry covariance structure, was used with site as a random factor (PROC MIXED, SAS 9.4, SAS Institute, Inc., Cary, NC, USA). Site was originally a main effect and the analysis resulted in expected season by site interactions. However, there was no biochar main effect or biochar by site interaction, so presented models use site as a random factor to account for the effect of site and expand the scope of inference from specific study locations to the area represented by randomly selected locations within the region. Soil moisture was used as a covariate in the CO_2 efflux model because of improvements to model fit.

The effect of biochar on soil C content and annual tree growth rates were analyzed using a one-way mixed effects model (PROC MIXED), with site as a random factor using

TABLE 4 Repeated measures analysis *F* statistic and *p*-values of the measured effects of biochar treatment (T) and season (S), for CO_2 efflux, CH_4 uptake, soil moisture, and soil temperature from all forest biochar-amended sites in 2015. Boldface indicates significance at $p \leq 0.05$

	CO_2 efflux		CH_4 uptake		Soil moisture		Soil temperature		C content		Diameter growth	
Effect	F^a	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
T	0.72	0.53	0.53	0.59	0.62	0.58	0.58	0.59	7.82	**0.03**	0.17	0.84
S	85.51	**<0.01**	8.79	**<0.01**	144.75	**<0.01**	625.49	**<0.01**				
T x S	0.24	0.91	1.83	0.12	1.22	0.31	2.03	0.09				

[a]F statistics (*F*) and *p*-values (*p*).

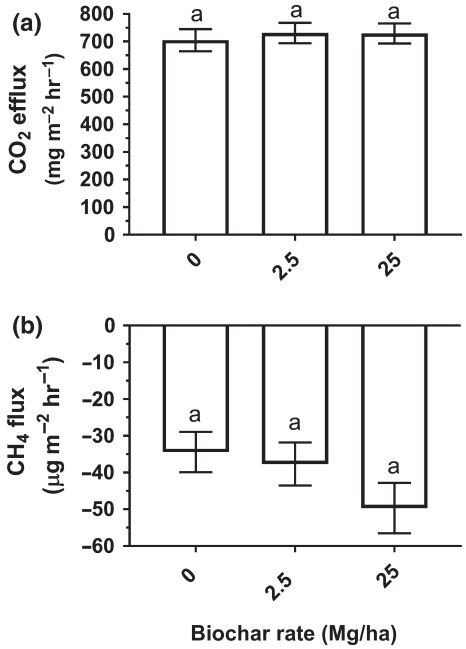

FIGURE 2 Average carbon dioxide (mg CO_2 m^{-2} hr^{-1}) (a) and methane flux (μg CH_4 m^{-2} hr^{-1}) (b) rates from all study sites by biochar treatment rate. Bars represent standard error ($n = 75$). Columns with same letters are not statistically different ($p > 0.05$)

Type III test of fixed effects to account for unbalanced design. Differences between biochar rates were considered significant at $p \leq 0.05$. If a significant effect was found, Tukey–Kramer tests were performed for multiple comparisons. If normality and equal variance assumptions for analysis of variance were not met, the data were transformed using the Box-Cox method with PROC TRANSREG (SAS 9.4). Data for CO_2 efflux, CH_4 uptake, and C storage were all transformed. Optimal covariate structure for repeated measures was selected by corrected Akaike information criteria (AICC).

3 | RESULTS

3.1 | Soil GHG emissions

Carbon dioxide efflux and CH_4 uptake (negative flux) varied across seasons and by location but were not affected by biochar treatment rate (Table 4, Figure 2 and Supporting Information Figure S2). Nitrous oxide flux was not detected in this study. Summer CO_2 efflux rate was 57% higher than spring and 63% higher than fall rates (Supporting Information Figure S2a) while CH_4 uptake was 45%–61% higher (i.e., CH_4 flux was more negative) in summer than in spring and fall (Supporting Information Figure S2b). The Umpqua site had the highest CO_2 efflux rate while UIEF had the lowest (Supporting Information Table S2). Umpqua and Purdue Creek had the greatest CH_4 uptake while Swift Creek had the lowest (Supporting Information Table S2).

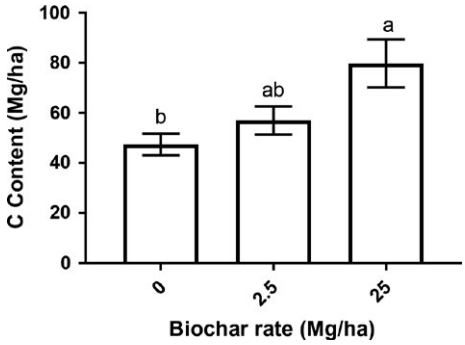

FIGURE 3 Average mineral soil C content (Mg/ha) in the 0–15 cm sampling depth for each biochar application rate at the end of 1–6 years for all study sites. The C content was calculated based on fine fraction bulk density. Bars represent standard error ($n = 32$, 0 Mg/ha; $n = 27$, 2.5 Mg/ha; $n = 16$, 25 Mg/ha). Points with same letters are not statistically different ($p > 0.05$)

3.2 | Soil moisture and temperature

As expected, soil moisture and temperature varied with season (Table 4, Supporting Information Figure S3). Soils were dry and warm in the summer. Soils were wet and cooler in the spring and fall. Purdue Creek was wettest, and Swift Creek was driest (Supporting Information Table S2). UIEF and Umpqua were the warmest while Pitwood was coolest during the sampling period (Supporting Information Table S2).

3.3 | Soil C content

The application of biochar at all rates increased soil C content (Table 4, Figure 3). Forest soils amended with biochar at the rate of 25 Mg/ha had 41% more C than forest soils with 0 Mg/ha biochar added. Purdue Creek soil had the highest C content while Umpqua had the lowest C content (Supporting Information Table S2).

3.4 | Tree diameter growth

For all sites, tree diameter growth was not affected by biochar application rate ($p = 0.84$). Average tree diameter growth was 0.95 ± 0.17 cm/year for trees grown in 0 Mg biochar/ha, 0.93 ± 0.18 cm/year for trees grown in 2.5 Mg biochar/ha, and 1.02 ± 0.19 cm/year for trees grown in 25 Mg biochar/ha. The greatest tree diameter growth occurred at Umpqua, and the slowest tree diameter growth occurred at UIEF (Supporting Information Table S2).

4 | DISCUSSION

4.1 | Soil GHG emissions

For all study sites, application of biochar had no influence on forest soil GHG fluxes. The study sites

represent a range of soil temperature and moisture conditions throughout western US forests. The biochar used was from a variety of feedstocks and pyrolysis conditions. The time since application ranged over several years. Yet, in no case did we see an effect on GHG flux. Consequently, we conclude that biochar has little influence on GHG fluxes when applied to western US forest soils. A recent meta-analysis considering many soil types found that biochar amendments caused no changes in CO_2 efflux in field trials, but biochar amendment did increase field-measured CH_4 flux into the atmosphere by 25.4% (He et al., 2017). However, when considering field-measured forest soils, biochar amendment had no effect on temperate hardwood forest GHG flux (Sackett et al., 2015) or on CO_2 efflux in a subtropical forest (Zhou et al., 2017). Laboratory studies of forest soils show CO_2 and N_2O efflux increased and CH_4 uptake decreased with biochar amendment (Hawthorne et al., 2017; Johnson, Webster, Jassal, Hawthorne, & Black, 2017; Mitchell, Simpson, Soong, & Simpson, 2015). However, we are not aware of any study comparing laboratory responses to field responses. Although laboratory studies are highly controlled and easily replicated, they do not necessarily reflect actual field conditions or variability and are usually completed on a much smaller scale compared to operational level field experiments. In addition, operational level field experiments provide more reliable and practical results for field scale biochar application for climate change mitigation (Song, Pan, Zhang, Zhang, & Wang, 2016). Our results, combined with other field-based forest reports confirm biochar increases or has no effect on CO_2 or N_2O efflux and CH_4 uptake, while decreases have not yet been reported.

The lack of GHG flux differences between amended and non-amended biochar soils in this study could be due to the timing of measurements in relation to biochar application. Biochar can cause a short-term increase in CO_2 efflux due to microbial response to biochar's labile C and other nutrients (Ameloot, Graber, Verheijen, & Neve, 2013). In a Chinese chestnut plantation, CO_2 efflux increased within one month of biochar amendment relative to unamended controls, but not thereafter (Wang et al., 2014). Since we did not measure GHG flux for months or years after amendment, any initial increase in CO_2 efflux due to microbial oxidation of labile C occurring in the biochar-amended soils may have been undetected. Our results indicate that biochar does not cause increased decomposition of native organic matter, as suggested from litter bag studies (Wardle, Nilsson, & Zackrisson, 2008).

Biochar application method may affect differences in GHG flux results between forest and agricultural biochar amendment studies. Mixing biochar into the soil surface can increase water holding capacity (Mukherjee & Lal, 2013), which can affect CH_4 uptake (Le Mer & Roger, 2001). We applied biochar to the soil surface because it is not practical to incorporate biochar into forest soil (Page-Dumroese, Coleman et al., 2016) and such disturbance is not desirable. When our GHG sampling occurred, the biochar was still at the soil surface, although it was beginning to mix into the soil profile. Over time, through natural soil mixing (Gavin, 2003), biochar will be further integrated into the soil profile. Incorporating biochar into the soil profile is much easier in agricultural settings due to site conditions, available equipment, and common acceptance of tilling practices. Such disturbances are considered detrimental to forest ecosystem functioning as organic matter supports soil nutrient cycling, water availability, gas exchange, and biological diversity (Binkley & Fisher, 2013).

Soil texture can also cause GHG flux to vary. When considering soil texture and biochar amendment, a recent meta-analysis found soil texture to significantly affect GHG flux, but the magnitude and direction depended on the GHG (He et al., 2017). Biochar causes CO_2 efflux to increase, except in fine textured soil. In addition, biochar decreases CH_4 efflux only in coarse-textured soil, and biochar generally decreases N_2O flux in all soil textures (He et al., 2017). Although we did not analyze soil texture directly, we do see trends of relative CO_2 efflux rate and relative CH_4 uptake rate varying due to biochar amendment dependent on soil texture (Supporting Information Figure S4). For example, relative CO_2 efflux rates increased in coarse loamy soil amended with 25 Mg/ha biochar, but relative CO_2 efflux rates decreased when coarse loamy soil was only amended with 2.5 Mg/ha biochar (Supporting Information Figure S4). Even though our study and the meta-analysis by He et al. (2017) were not set up to look at soil texture directly, soil texture could be important in GHG flux in biochar-amended soils and should be measured in future studies.

Nitrous oxide flux was not detected in this study due to low N availability. Nitrogen is often limited in western United States coniferous forests (Coleman, Shaw, Kimsey, & Moore, 2014), which leads to rapid immobilization, low nitrification (Stark & Hart, 1997), and consequently limits the main source of N_2O emissions in well-aerated soil. In aerated soils, nitrous oxide emissions can occur as a byproduct of nitrification (Firestone & Davidson, 1989). Nitrous oxide emissions can also occur in anaerobic microsites or in soils with excessive water content from anaerobic denitrification when nitrate is used as an alternative electron acceptor in the absence of oxygen (Robertson & Groffman, 2015). The lack of N_2O emissions in our study indicates that there was insignificant anaerobic-microsite denitrification. We did not consider biochar amendments in water-saturated soils, so it is possible that biochar may limit N_2O emissions under those conditions, although that is also expected to be limited without significant nitrate supply.

4.2 | Soil C content

Biochar amendment increased forest soil C content from 17% to 41% (Figure 3). Previous work showed that adding biochar to soil enhances soil C storage (Zimmerman, Gao, & Ahn, 2011). For instance, forest soil amended with 16 and 32 Mg/ha biochar increased soil organic carbon by 20% or 40% (Bamminger, Marschner, & Jueschke, 2014). Biochar adds directly to long-term storage pools. Biochar has a large amount of recalcitrant C, which can remain in the soils for hundreds of years (Wang et al., 2016). Long residence times in the soil makes biochar a useful tool for C sequestration (Lehmann, Gaunt, & Rondon, 2006; Wang et al., 2016) without causing harm to western US forests.

4.3 | Tree diameter growth

Tree diameter growth was unaffected by biochar amendment after 1–6 years. A meta-analysis found that there is a potential for an average of a 41% increase in tree growth response to biochar (Thomas & Gale, 2015), but most studies in the meta-analysis were from short-term tree seedling pot studies, not forest site investigations. In an 18-month pot study using highly degraded urban soils and small (1–2 cm root collar diameter), bare root seedlings of *Acer saccharum* and *Gleditsia triacanthos*, Scharenbroch, Meza, Catania, and Fite (2013) found that across species and soil types, biomass increased 44% with biochar applied at 25 Mg/ha compared to control pots (no biochar), but this was an 18-month pot study. Sarauer and Coleman (2018a) found biochar amended to peat-based substrates at 25% and 50% v/v to reduce Douglas-fir seedling growth due to pH increasing to over 7, well above the favorable pH range for conifers of 5.2–6.2 (Binkley & Fisher, 2013). The study reported here was a long-term field trial using established trees. The older age of the trees could result in slower responses to biochar amendment. In addition, the soils at each site are buffered by the forest floor, inherent soil organic matter, and active microbial communities which buffer pH and resulted in no changes in pH after 6 years (Magdoff & Bartlett, 1985).

Previous field-based studies in tropical or subtropical forests found soils amended with biochar increased (Lin et al., 2017; Sovu, Savadago, & Oden, 2012) or had no effect (de Farias et al., 2016; Lin et al., 2017) on tree growth. Mixed wood ash applied to a *Pinus radiata* planation caused an increase in tree growth three years after application in a Mediterranean climate (Omil, Pineiro, & Merino, 2013). However, in temperate regions, biochar can have small or even negative effects on soil properties and crop responses (Jeffery, Verheijen, Velde, & Bastos, 2011; Jones, Rousk, Edwards-Jones, DeLuca, & Murphy, 2012). The lack of biochar enhancement to tree growth in our study could be from N limitation or possibly because of low amounts of soil phenolics or other growth-inhibitory substances in temperate soils (Thomas & Gale, 2015). Another possible reason for a lack of a biochar effect on tree growth observed in this study could be due to the particular tree species investigated. Angiosperm seedlings respond better to biochar than gymnosperm seedlings (Pluchon, Gundale, Nilsson, Kardol, & Wardle, 2014) possibly due to angiosperm seedlings responding more to soil fertility than gymnosperm seedlings (Bond, 1989; Coomes et al., 2005) or poor adaptation to increased pH by gymnosperms (Sarauer & Coleman, 2018a). Thomas and Gale (2015) also suggest that the reduced response from conifers could be due to lower nutrient uptake rates and adaptation to surviving in acid soils causing conifers to be resource-conservative compared to angiosperms. In addition, these forests are found in fire-adapted ecosystems which have a high inherent black C component (DeLuca & Aplet, 2008) and therefore, the amount of biochar added may not be significant considering abundant wildfire-produced charcoal already in the mineral soil and forest floor. It should be noted that we did not determine the baseline pyrogenic C deposited from past wildfires, so we are limited to the soil C measured after biochar application. Western USA forests, mineral soil (10-30 cm depth) can contain over 100/Mgha^{-1} C, but C deposited from wildfire can vary across landscapes (Chen & Shrestha, 2012) so lacking baseline pyrogenic C measurements might be a limitation. However, C storage changes observed in relation to control plots clearly show a stepwise increase in stored carbon and therefore, biochar amendment in this study did increase C storage. In addition, since the biochar was surface applied, it took more than three years for the high rate biochar to move through the forest floor and into the mineral soil. For example, on the Umpqua site, biochar was still visible on the surface of the forest floor after 4 years. This suggests that the biochar may not yet be significantly influencing mineral soil chemical and physical properties. Changes associated with biochar may occur over time as the biochar moves down the soil profile. In addition, most of our soils were Andisols or soils with Andic properties. Biochar may not have a significant effect on Andisols because of the porous structure of volcanic pumice and ash leading to an increase in water holding capacity and therefore increased tree productivity (Meurisse, Robbie, Niehoff, & Ford, 1991).

4.4 | Site differences

The sites used in this study were very different (Supporting Information Table S2) and accounted for much of the variation in our study. The study sites varied in latitude, elevation, mean annual precipitation, mean annual temperature, tree age, and soil characteristics, all of which affected the variables measured in our study. Even though the sites were very different, we found within-site seasonal differences, but not biochar differences. The variation accounted for by site and season improved the sensitivity for the effects of biochar. The regional distribution of study sites and including site as random effect in the statistical model expanded the scope of our study to encompass forests throughout the Pacific and Rocky Mountain western USA.

The sites also received biochar from different sources. In particular, the Idaho sites, Pitwood, Purdue Creek, and UIEF were amended with the high ash content biochar, Evergreen Forest Products Biochar. Previous research investigating traditional biochar and high-ash char (also called wood-ash) has found the two products to effect plant growth, soil properties, and biogeochemical processes differently. For example, Noyce et al., 2017 performed a short-term microcosm study with acidic Ontario soils and found wood ash and biochar to have differing effects on soil pH, nutrient availability, and 12 week seedling growth of sugar maple (*Acer saccharum*) and red pine (*Pinus resinosa*). However, in this study we found biochar source, whether a pyrolysis biochar (e.g. the biochar used at Umpqua or Swift Creek) or a high wood ash biochar (e.g. the Evergreen Forest Products biochar) to not affect soil GHG flux or tree growth in western USA forests.

4.5 | Implications

Overcrowded forests may need to be thinned to reduce wildfire and susceptibility to insects and drought. When forests are thinned, the unmerchantable biomass is often burned in slash piles which is both wasteful and a source of smoke and particulate pollution. Therefore, a possible use of woody biomass residues is conversion to biochar and returning it to the forest soil for C mitigation and to potentially improve soil properties. We found that applications of biochar to forest sites did not have long-term impacts on GHG emissions and tree growth; therefore, biochar was not detrimental to western USA forests. At the same time, amended soils contained more C than soils without biochar. Based on these results, applying biochar to forest soils can be a climate mitigation tool that will sequester C and will not adversely affect soil GHG emissions or conifer diameter growth in the western USA.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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