Carbon market facilitated afforestation of U.S. grasslands could yield a net carbon loss

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1. Introduction

There is increasing attention on the role of afforestation to provide a "natural solution" to climate change. However, the ability of afforested lands to sequester carbon is moderated by pre-afforestation land use and soil carbon response. Past land use, initial soil carbon stocks, soil texture, and disturbance regime contribute high variability and uncertainty around the efficacy of afforestation to store ecosystem carbon over the medium to long term (multiple decades to a century). This is especially true in rangelands of the U.S. Great Plains, where soil carbon gains in response to afforestation are low or even negative, and carbon gains in aboveground woody vegetation are vulnerable to wildfires. Critically, grasslands are the land use type most vulnerable to afforestation facilitated by carbon markets due to the low marginal costs of grassland conversion and low opportunity costs relative to, say, cropland (Nielson et al., 2014). Further, afforestation in an era of rapidly changing disturbance regimes could drive the creation of novel ecosystems that store less total (above and below ground) carbon than the pre-afforested ecosystem. Avoiding such unwanted outcomes while proceeding with afforestation as a natural climate solution requires an understanding of the ecological components that regulate carbon storage, the economic drivers behind afforestation, and the ecological thresholds of afforested lands (i.e., tipping points suddenly crossed in response to relatively incremental change).

A. Afforestation and Soil Carbon Storage. Soil carbon underpins a variety of ecosystem services including regulation of the atmosphere and climate, carbon sequestration, primary (including agricultural) production, waste processing, decomposition, nutrient conservation, water purification, erosion control, and disease mitigation (Wall et al., 2004; Bardgett, 2005; de Deyn and Van Der Putten, 2005; Wall et al., 2015). Soil carbon (C) can be defined as organic matter smaller than 2 mm with tight associations in the soil matrix (Stewart et al., 2007). Soil C is a larger, and more stable pool of carbon than aboveground biomass C: soil C stores are greater than that of all living biomass and atmospheric CO_2 combined (Jobbagy and Jackson, 2000). This vast store of carbon does not predictably respond to afforestation. A metaanlysis conducted by Post and Kwan (2000) reported that total soil C varied from -0.141 to 0.617 Mg C $ha^{-1}yr^{-1}$ in response to afforestation. Another metaanlysis reported 25% greater SOC gains in crop versus pasture land (Guo and Gifford, 2002) and a review from Paul et al. (2002) reported increased SOC in afforested cropland and a loss of SOC from afforested pastureland.

The complexity of soil systems paired with a lack of long term soil C specific datasets make it difficult to ascertain the mechanisms driving varied soil C response to afforestation. Woody species identity, soil texture, past land use and cover, depth of soil profile, soil texture, and time since afforestation likely play a role in the strength and direction of the relationship (McKinley at el., 2011). The carbon saturation hypothesis offers one way to bound this complexity. The C-saturation theory uses non-linear, second order kinetics to explain the decline in soil C accumulation rates as the upper limit of soil C storage is reached (Six et al., 2002, Stewart et al., 2007; Figure 1). This is in contrast to the first order kinetics used in many global C models that describe soil C stores as proportional to the rate of inputs without an upper limit. While the first order kinetics model fits soil C storage efficiency (i.e., ratio of soil C increase to C inputs) in systems with low C, it performs poorly on soils with high C inputs and storage, such as those of historical grasslands. This C saturation model better fits long term soil carbon storage in which incremental gains of soil C decline as the rate of C inputs increases, i.e. soils become less efficient at storing C as they reach saturation (Six et al., 2002; Stewart et al., 2007).

The saturation point of soil C is likely the result of physicochemical processes of adsorption of C to clay surfaces (Hassink 1996, 1997), protection in aggregates, and the biochemical complexity of the C inputs (Baldock and Skjemstad 2000). It includes both the protected (on clay surfaces and in aggregates) and unprotected soil C pools to store inputs. These

Significance Statement

If we submit to PNAS, here is the 120-word maximum statement about the significance of the research paper written at a level understandable to an undergraduate educated scientist outside their field of speciality. The primary goal of the Significance Statement is to explain the relevance of the work in broad context to a broad readership. The Significance Statement appears in the paper itself and is required for all research papers.

H.E.B. conceived of the idea and wrote the manuscript. C.R.A. and D.T. contributed equally to this work, providing overall framing and writing the synthesis. C.H.B. and C.K. contributed to the writing of the manuscript. J.G. and A.J.P. provided the analyses and contributed to writing the manuscript

The authors declare no conflict of interest.

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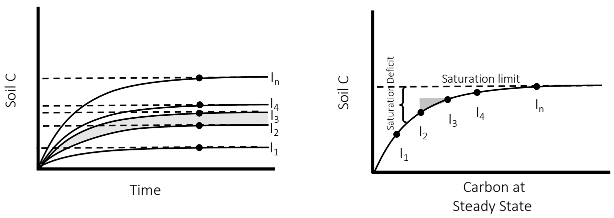

Fig. 1. Soil carbon (C) dynamics described by the C-saturation hypothesis, showing the effect of different carbon input levels (I_x) on soil C over time (left panel), and at steady state (i.e., inputs and release in equilibrium) (right panel). As inputs increase, storage capacity and steady state carbon storage decrease. The greyed area shows gains in soil carbon when inputs increase. Adapted from Stewart et al., 2007.

pools in turn are modified by land management, plant cover, belowground diversity, and climate. Saturation equilibria have been detected to shift upwards in some systems following nutrient fertilization (e.g., Russell, 1960), and changes in plant diversity and the quality of litter inputs (Follett et al., 1997), indicating that saturation limits can be dynamic.

It is unknown how C saturation will be affected by afforestation. Notably, grassland soils are generally more C rich than their forested counterparts. Global grassland soils, on average, store an estimated average 331 Mg C ha^{-1} (Schlesinger, 1977; Conant, 2001) versus an average 96 Mg C ha^{-1} in global forest soils (Lal, 2005). Grasslands store 10-30% of the world's soil carbon (Anderson, 1991; Eswaran et al., 1993), or 200-420 Gt C (Ojima et al., 1993c; Scurlock and Hall, 1998; Batjes, 1999). This difference is understood to be the result of higher aboveground and fine root litter contributions to soil and climatic conditions that favor storage over loss of soil C in grasslands. Whether grasslands and forests have intrinsically different soil carbon saturation limits and how dynamic those limits are in response to land cover provides insight into the implications of afforestation of grasslands for C sequestration. Specifically, if forest cover has intrinsically lower soil carbon storage capacity, a loss of aboveground afforested biomass could yield a net loss of carbon from afforested grasslands (Figure 2). Poor understanding of the underlying mechanisms of soil C saturation make it difficult to predict how saturation equilibria might be altered with grassland conversion to forest and whether any downshifts in saturation equilibria are reversible with grassland restoration. Importantly, if forest cover *increases* soil carbon in former grasslands, the likely mechanism is via increased inputs rather than higher saturation limits. If instead forested cover reduces the overall sequestration potential, a larger amount of soil carbon than explained by reduced inputs along could be lost due to the second order decay dynamics that describes slowing storage rates as soil reaches a new, lower saturation (Figure 2).

B. Afforestation, Aboveground Carbon, and Wildfires. Aboveground woody biomass C can be vulnerable to permanent losses from wildfires when the native disturbance regime is lost. Historical fire-adapted forests experienced periodic fires that preceded stand regrowth. The fires released large amounts of C-CO_2, with dead aboveground necromass slowly releasing more C over decades at it decomposed. At the same time, post-fire regrowth of forest plants sequestered carbon, allowing fire adapted forests to recover carbon lost to fire over the fire-regrowth period of their life cycle (Kashian et al., 2006; Rothsteian et al., 2004; McKinley et al., 2011). When such fire-regrowth successionary cycles are distributed through space and time, it moderates the overall fire-driven carbon flux from multiple fire adapted forest ecosystems (Harmon, 2001). An average fire in the continental US releases 213 ($\pm$ 50 std. dev.) Tg CO_2 yr^{-1} of carbon, and total CO_2 losses from forest fires in the US are equivalent to 4–6% of total anthropogenic emissions. The typical forest fire-regrowth successionary cycle is on a scale of multiple decades to a century, so if 1) fire return intervals overtake regrowth periods, or 2) fires reach intensities high enough to stymie regrowth, losses will outpace inputs.

In the modern Great Plains, the historical fire disturbance regime (i.e., intensity, return interval, area burned, etc) is altered. Native juniper (Juniperus virginiata) patches once relegated to rocky outcroppings, river corridors, and transient outbreaks into the prairie matrix (Biggs, 2002) have expanded into roughly 7 million hectares of former Great Plains grasslands with no signs of slowing. This is attributable to direct fire suppression and land use change (Engle and Kulbeth 1992; Briggs et al. 2002). Frequent, low intensity grassland fires occasionally spreading to (and eliminating) juniper patches in are being replaced by large, high intensity forest fires that sweep across large, connected expanses of juniper forest (Donavan et al., XXXX). It is estimated that tallgrass prairie burned every 3-5 years, likely during the hot, dry growing season (Wright and Bailey 1982; Knapp and Seastedt 1998). Such fires were typically intense enough to remove fire-sensitive woody plants like juniper from the landscape, and favored the dominance of fire-tolerant graminoid and forb species (Archer 1994). When successionary cycles of fire and regrowth (also known as "adaptive cycles", Allen et al. [XXXX]) are inadvertently synchronized, the system is primed for a large release of carbon from a catastrophic wildfire season (Smithwick et al., 2007). This synchronization is exemplified by surging western United States (US) wildfires in recent decades, which are likely exacerbated by climate change.

As humans continue attempting to suppress fires in the Great Plains, closed canopy juniper forests contain a fuel load that makes fire difficult to suppress. While grassland fires typically have flame lengths between 0.1 to 3.4 m (Finney et al. 2011; Twidwell et al. 2016), it is not uncommon for crown fires on juniper forests to have flame lengths well over 14 m (Twidwell 2012). The US Forest Service specifies that wildfires with flame lengths over 3.4 m are unlikely to be suppressed. Juniper forests in the North American Great Plains are therefore highly vulnerable to uncontrollable wildfires (Andrews and Rothermel 1982; Engle et al. 1996; Twidwell et al. 2013b; Donavan et al., 2017). This emphasizes how short lived aboveground carbon gains may be in Great Plains grasslands afforested with juniper. There is a magnitude difference of ecosystem carbon lost from fires when a grassland with an estimated 1660 kg aboveground C ha^{-1} (Knapp et al., 1998) versus a juniper woodland with 64,090 kg aboveground C ha^{-1} (Norris et al. 2001).

Soil carbon, by contrast, does not have a predictable strength or direction relationship to fire. The influence of fire on soils is likely mediated through aboveground plant community shifts, and through charcoal inputs. Rice (2000) reported an increase in soil C of 2.2 Mg ha^{-1} over a ten year period of annual burning and grazing of tallgrass prairie, al-

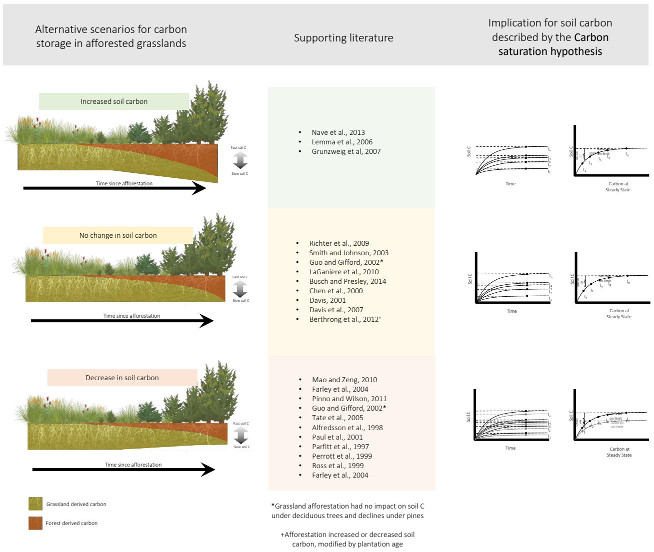

Fig. 2. Alternate scenarios of afforestation on soil carbon stocks over time, with supporting literature, and the predicted change in soil carbon according to the carbon saturation hypothesis. The scenarios include gains in soil carbon (highlighted in green), no affect on soil carbon (highlighted in yellow) and declines in soil carbon (highlighted in red) in response to afforestation. Only pasture and rangeland (i.e., cropland cases omitted) afforestation cases are included in the supporting literature examples.

though the relative contribution and/or interactions of each process is unknown. In many grasslands exposed to frequent fire, soil C is derived from charcoal inputs (also referred to as black C or biochar) (Skjemstad et al., 1996). With its a high mean residence time, charcoal is estimated to comprise up to 35% of global soil C (e.g. Glaser et al. 2000; Schmidt et al. 2002; Ding et al. 2012), although the local mean residence time of charcoal varies from a half-life of less than fifty to over 100 years (Bird et al., 1999; Zimmerman, 2010). While charcoal inputs are lower in grassland versus forest fires (a < 3 versus a 4-5% conversion rate of biomass to charcoal; Forbes et al., 2006), it is unlikely that juniper fires will occur with enough frequency to lower soil C turnover time (i.e., increase the size of the slow pool through charcoal inputs). Additionally, the high fuel load of juniper forests and warm, hot dry weather of the Great Plains is likely to produce high combustion efficiency (i.e., smoldering) that converts biomass to ash, which is low in organic carbon (Forbes et al., 2006).

While aboveground afforestation carbon gains in the Great Plains are vulnerable to loss from wildfire, soil carbon changes could offset or amplify this loss. A readily apparent implication of grassland-juniper afforestation is an increase in aboveground carbon (Norris et al., 2001; McKinley, 2006; McKinley et al., 2008; Busch and Presley, 2014; Mellor et al., 2013). Juniper has inherently high growth rates relative to prairie graminoids and forbs, with much of this growth allocated to aboveground, perennial biomass. In contrast, grassland species allocate a larger share of their annual production to the soil surface as litter (Chapin, 1980; Vitousek, 2004; McKinley et al., 2008) and to the soil as root biomass turnover (Kuzyakov and Domanski, 2000). Aboveground carbon (C) in 50 year old juniper stands in the Flint Hills of the central US Great Plains were reported to contain roughly 64,090 kg C $ha^{-}1$, with 52% of total ecosystem carbon stored in the soil (Norris et al., 2001; McKinley, 2006). By comparison, adjacent tallgrass prairie cover holds only an estimated 1660 kg aboveground C $ha^{-}1$ at peak growing season, but with 96% of total ecosystems carbon stored belowground (Knapp et al., 1998). If afforestation decreases soil carbon storage, and allocations are shifted towards aboveground carbon vulnerable to fire, afforestation could lead to a net loss of carbon over a timescale of decades to a century.

C. Novel stable states/resilience section. Written by Craig/Dirac. Figure showing range->moderate juniper -fire-> range AND range->heavy juniper -fire-> ??

D. Carbon market facilitated afforestation. There is growing interest around afforestation-derived carbon sequestration as a plausible supply of carbon credits to possible carbon markets (e.g., Moulton and Richards, Adams et al., 1993 Nielson et al., 2014). Nielson et al. (2011) provided detailed cost estimates for carbon storage through afforestation, assessing county level costs for cropland, pasture, and rangeland afforestation, accounting for biophysical limits to forest growth rate and opportunity costs (i.e., lost returns from agricultural production of cropland afforestation). In the Great Plains, the cost of tree establishment ranges from \$1-6,000 per acre; showing higher variability in cost than the eastern (\$1-100 per acre) and western US (\$300-6,000), largely due to biophysical drivers and opportunity costs of converting otherwise productive cropland (Nielson et al., 2014). However, with juniper encroachment the actual cost across of tree establishment in much of the Great Plains is \$0, further incentivizing grassland afforestation as a lucrative endeavor.

Indeed, if a carbon market provided a mechanism to buy credits, rangeland would be most vulnerable to conversion. Nielson et al. (2011) showed that C priced at \$50 per metric ton would facilitate the afforestation of 118.4 million acres of rangeland, 48.2 million acres of pasture land, and 79.2 million acres of cropland. In this scenario, nearly half of carbon sequestered from afforested lands (93.7 of the 200 M tons) is attributed to rangeland conversion -a pool of carbon vulnerable to combustion and release back to the atmosphere.

Here we (insert a brief summary describing the mapping approach).

2. Results

words words

3. Synthesis

TO BE WRITTEN Points: 1. The risk of afforesting grasslands in non-stationary systems. Discuss time frame for carbon storage in juniper woodlands vulnerable to fire cycles (50-200 yrs?) vis-a-vis policy time scales (4-10 years?) and soil carbon response (immediate to millenia). 2. The large juniper forests and resulting fires are without analogue and could lead to alternative stable states that cannot support regrowth -or contributions to soil C stores 3. Why carbon markets are more complex that initially understood: they offer an attractive way for the market to mitigate climate sure, but there are some

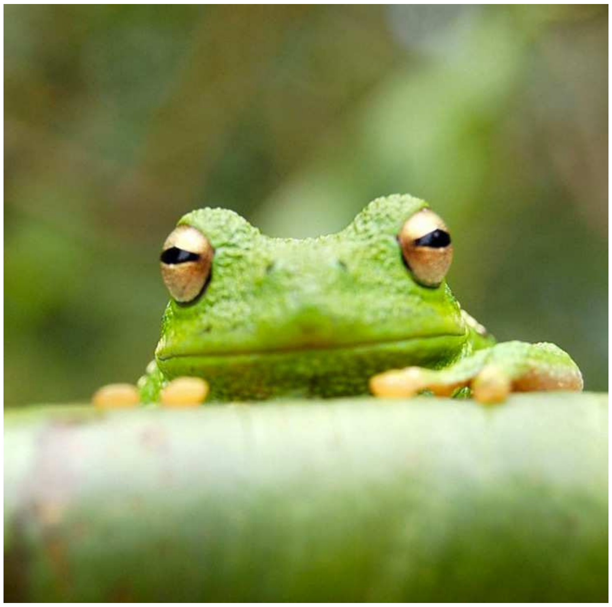

Fig. 3. This image is a place holder for the map.

Table 1. Table in case we need it; placeholder

Species	CBS	CV	G3
1. Acetaldehyde	0.0	0.0	0.0
2. Vinyl alcohol	9.1	9.6	13.5
3. Hydroxyethylidene	50.8	51.2	54.0

big hidden costs (like this) that need to be ironed out before any type of market starts valuing afforestation.

4. Conclusion

Need to focus on preventing deforestation, managing/conserving existing native forest and grasslands for carbon storage, and mitigating emissions. Proceed carefully with carbon markets. No easy fix to a complex social, economic, and ecological issue.

5. References

Materials and Methods

Please describe your materials and methods here. This can be more than one paragraph, and may contain subsections and equations as required. Authors should include a statement in the methods section describing how readers will be able to access the data in the paper.

Subsection for Method. Example text for subsection.

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