Highlights

* natural climate solutions include managing forests to store carbon
* payments for carbon could lead to 61.2 million acres of additional afforested US rangeland
* payments for forestry carbon offsets could undermine other ecosystem services including fire suppression
* offset generation from afforestation may lead to a net loss of carbon
* offset methodologies can mitigate the tradeoffs of afforestation

The limited value of carbon benefits from grassland afforestation in the

U.S. Great Plains

Hannah E. Birge*a,b*,\*, Christine H. Bielski*b*, Jacob C. Gellman*c*, Dat T. Ha*a*, Clare Kazanski*a*, Andrew J. Plantinga*c*, Craig R. Allen*d* and Dirac Twidwell*b*

*aThe Nature Conservancy, 4245 Fairfax Dr 100, Arlington, VA 22203*

*bDepartment of Agronomy and Horticulture, University of Nebraska-Lincoln, Lincoln, NE, 58583*

*cBren School of Environmental Science & Management, University of California, Santa Barbara, CA 93106*

*dU.S. Geological Survey, Nebraska Cooperative Fish and Wildlife Research Unit and The School of Natural Resources, University of Nebraska, Lincoln, NE, USA*

\*Corresponding author

A R T I C L E I N F O

*Keywords*: ecosystem services, carbon offsets, natural climate solutions, grasslands, soil carbon

**Abstract**

In the United States, enhancing forest growth on native grasslands may initially appear as a cost-effective pathway for storing additional ecosystem carbon and generating carbon offsets needed by governments, companies, and other entities to reach their decarbonization targets. Yet, the social and ecological costs of grassland afforestation in the United States Great Plains includes increased vulnerability to wildfires, loss of critical habitat and other grassland ecosystem services, and even the potential for lower net ecosystem carbon over the long term (10+ year), especially if afforestation limits carbon storage equilibria in the soil. Failing to account for those costs during offset generation could lead to the afforestation of an additional 61.2 million acres of grasslands in the United States (U.S.) Great Plains under a scenario of carbon credits priced at $50 per metric ton CO2-e. Fortunately, methodologies from carbon registries, like the Climate Action Reserve, can be applied to account for both the benefits and costs of afforestation, thereby likely lowering its value as a natural climate solution. By including these considerations in the scoping phase of project development, carbon offset credit registries and project developers can avoid afforestation scenarios that undermine their commitment to generating durable carbon that incur no net harm on the social-ecological system.

# Background

Land management activities that enhance carbon storage in ecosystems represent an important pathway for mitigating global climate change. These so called "natural climate solutions" (NCS) ([Griscom et al.](#_bookmark15) [[2017](#_bookmark15)]) include managing forests, pastures, wetlands, and croplands to increase carbon fixation and retention in plants and soils. Avoided deforestation, reforestation of formerly forested lands, and afforestation of non-forest lands are all natural climate solutions that leverage the carbon storage benefits of forests ([Plantinga and](#_bookmark52) [Wu](#_bookmark52) [[2006](#_bookmark52)], [Forster et al.](#_bookmark11) [[2021](#_bookmark11)]). Yet afforestation of grasslands is distinct from the other two categories of forestation due to the well-documented loss of ecosystem service provisioning that occurs when trees invade grasslands. Those lost ecosystem services include fire suppression ([Donovan et al.](#_bookmark7) [[2017](#_bookmark7)]), long term (10+ year) carbon storage ([Jackson](#_bookmark24) [et al.](#_bookmark24) [[2002](#_bookmark24)]), habitat provisioning ([Wilcox et al.](#_bookmark47) [[2022](#_bookmark47)]), water availability and quality ([Kishawi et al.](#_bookmark26) [[2023](#_bookmark26)]), forage production ([Morford et al.](#_bookmark42) [[2022](#_bookmark42)]), and recreational value ([Birge et al.](#_bookmark1) [[2016a](#_bookmark1)]).

However, carbon registries, like the Climate Action Reserve, have methodologies in place to safeguard against social and ecological costs of generating carbon offsets through NCS projects. There is, however, a lack of specificity in registry methodologies regarding how the costs of grassland afforestation in the United States (U.S.) Great Plains are operationalized in a project monitoring, reporting, and verification scheme. As more countries, corporations, and private citizens look to NCS as a solution for offsetting their greenhouse gas (GHG) emissions, it is essential that carbon registries provide explicit instructions around the accounting of climate and non-climate related ecosystem service trade-offs associated with afforestation of grassland systems.

# The ecology of grassland afforestation

Grasslands are a critically endangered global ecosystem: in the U.S. alone, is estimated that as little as 1 percent of the original grassland extent remained by the start of the 21st century ([Samson and Knopf](#_bookmark28) [[1994](#_bookmark28)]). Those grassland fragments provide myriad ecosystem services, including habitat for rare and federally endangered grassland species, forage production, nutrient and waste cycling, and carbon storage. When a grassland fragment is afforested, the loss of its ecosystem services can be sudden (i.e., nonlinear) and persistent (i.e., hysteretic) ([Twidwell et al.](#_bookmark38) [[2013](#_bookmark38)], [Birge et al.](#_bookmark1) [[2016a](#_bookmark1)]). The juniper (*Juniperus spp.*)invasion of U.S. Great Plains grasslands exemplifies this phenomenon. Junipers are woody plants native to the U.S. Great Plains but historically relegated to stream corridors and rocky outcroppings. Their expansion into grasslands is attributed to the suppression of grassland wildfires and the spread of propagules from planted tree stands (e.g., windbreaks) ([Elberg Nielsen et al.](#_bookmark8) [[2014](#_bookmark8)], [Twidwell et al.](#_bookmark40) [[2016](#_bookmark40)], [Donovan et al.](#_bookmark9) [[2018](#_bookmark9)]).

Reintroducing fire (e.g., controlled burns) and the use of mechanical removal can reverse afforestation during early juniper invasions ([Twidwell et al.](#_bookmark40) [[2016](#_bookmark40)], [Fogarty et al.](#_bookmark12) [[2022](#_bookmark12)]). At some threshold, however, invading junipers create a closed canopy forest vulnerable to uncontrolled wildfire and disproportionally more expensive to remove by mechanical means. Even when mechanical removal is successful, it can occur too late in the invasion process to restore grassland cover, i.e., the underlying self-reinforcing grassland ecosystem feedbacks are lost ([Angeler et al.](#_bookmark13) [[2015](#_bookmark13)]). When a mature juniper forest does burn, it often leaves behind standing remnant tree skeletons that interfere with light penetration, animal movement dynamics, and hydrology. Closed canopy juniper fires may also become hot enough to kill dormant grassland seeds in the soil. The threshold between reversibility and hysteresis in a grassland to juniper forest transition is dynamic and difficult to predict for any specific case ([Twidwell et al.](#_bookmark40) [[2016](#_bookmark40)]).

In addition to habitat loss, juniper afforestation of grasslands modifies nutrient cycling (e.g., nitrogen removal, [Reisinger](#_bookmark23) [et al.](#_bookmark23) [[2013](#_bookmark23)]) and ecosystem carbon storage. Global grassland soils store an estimated average 331 Mg C *ha*−1 ([Schlesinger](#_bookmark30) [[1977](#_bookmark30)], [Conant](#_bookmark4) [[2011](#_bookmark4)]) while forested soils store an estimated average 96 Mg C *ha*−1 ([Lal](#_bookmark32) [[2005](#_bookmark32)]). This could be attributed to grassland plants allocating a larger share of their annual production to the soil through root turnover and exudation (i.e., leaky roots; [Kuzyakov and Domanski](#_bookmark31) [[2000](#_bookmark31)]) and indirectly via the soil surface as dead plant tissue ([Chapin](#_bookmark5) [[1980](#_bookmark5)], [Vitousek](#_bookmark43) [[2004](#_bookmark43)], [McKinley et al.](#_bookmark34) [[2007](#_bookmark34)]). Junipers, by contrast, store much of their carbon in perennial aboveground biomass ([Norris et al.](#_bookmark50) [[2007](#_bookmark50)], [McKinley et al.](#_bookmark34) [[2007](#_bookmark34)], [Mckinley et al.](#_bookmark35) [[2011](#_bookmark35)], [Busch and Presley](#_bookmark6) [[2014](#_bookmark6)], [Mellor et al.](#_bookmark39) [[2013](#_bookmark39)]). In a fifty year old juniper stand in the U.S. Great Plains Flint Hills, aboveground carbon (C) was reported to contain roughly 64,090 kg C *ha*−1 with 52 percent of that total ecosystem carbon stored belowground (i.e., in the soil) ([Norris et al.](#_bookmark48) [[2001a](#_bookmark48),[b](#_bookmark49)], [McKinley et al.](#_bookmark34) [[2007](#_bookmark34)]). By contrast, an adjacent native grassland held an estimated 1660 kg aboveground C *ha*−1 at peak growing season, but had 96 percent of that total C stored belowground ([Knapp](#_bookmark29) [et al.](#_bookmark29) [[1998](#_bookmark29)]).

Whether afforestation alters the intrinsic capacity of the soil to store carbon is less understood, but essential from an offset generation perspective: if afforestation carbon shifts aboveground where it is vulnerable to fire and simultaneously reduces soil carbon storage, the effect could be a net loss of ecosystem carbon, depending on the probability of tree loss to wildfire. Soil C storage equilibrium is understood to be the result of physico-chemical adsorption of C to soil mineral surfaces ([Hassink and Whitmore](#_bookmark21) [[1997](#_bookmark21)]), physical protection of C in soil aggregates, the biochemical complexity of the C molecules themselves, and environmental limits on biological activity ([Baldock and](#_bookmark16) [Skjemstad](#_bookmark16) [[2000](#_bookmark16)], [Birge et al.](#_bookmark0) [[2015](#_bookmark0)]). Soil C saturation equilibria (fig. 1) have been detected to shift in some systems following changes in ecosystem processes (e.g., nutrient fertilization ([Russell](#_bookmark27) [[1960](#_bookmark27)]), indicating that soil C storage limits may shift in response to change in plant cover. If saturation equilibria do shift in response to afforestation, the reversibility of that shift following tree removal is unknown (fig. 2). A meta-analysis ([Post and Kwon](#_bookmark20) [[2000](#_bookmark20)]) showed soil C response to afforestation ranging from -0.141 to 0.617 Mg C *ha*−1*yr*−1 in response to afforestation, with much of the response variability predicted by tree species, soil texture, and environmental conditions. A different meta-analysis ([Guo](#_bookmark19) [and Gifford](#_bookmark19) [[2002](#_bookmark19)]) reported 25 percent greater soil C gains in afforested cropland versus grassland, and a review from Paul ([Paul et al.](#_bookmark51) [[2001](#_bookmark51)]) reported higher soil C in afforested cropland but a decreasein soil C when grasslands were afforested. [Barger et al.](#_bookmark17) ([2011](#_bookmark17)) reported similar variability in soil C response to afforestation and [Jackson et al.](#_bookmark24) [[2002](#_bookmark24)] reported that moisture levels change whether grassland afforestation leads to a net gain or loss of whole ecosystem carbon.

While the role of afforestation on soil carbon storage, and the impact of fire on soil is largely indirect, e.g., through charcoal inputs ([Skjemstad et al.](#_bookmark33) [[1996](#_bookmark33)]) and shifts in aboveground plant communities (e.g., [Rice et al.](#_bookmark25) [[2008](#_bookmark25)]), we do know that there is a magnitude difference in ecosystem carbon loss when a grassland (with an estimated 1660 kg aboveground C *ha*−1, [Knapp et al.](#_bookmark29) [[1998](#_bookmark29)]) versus a juniper woodland (with an estimated 64,090 kg aboveground C *ha*−1, [Norris et al.](#_bookmark49) [[2001b](#_bookmark49)]) burns.

1. **Afforestation and wildfire cost**

The allocation of ecosystem C to the aboveground following grassland afforestation leaves that C vulnerable to fire-related loss over the longer term (i.e., 10+ years) ([Harmon](#_bookmark22) [[2001](#_bookmark22)], [Donovan et al.](#_bookmark7) [[2017](#_bookmark7)]). In the U.S. Great Plains, mature juniper forest fires are disproportionality more difficult to repress relative to their grass fire counterparts and more likely to become uncontrolled wildfires that negatively impact infrastructure, private property, and public health ([Finney et al.](#_bookmark10) [[2011](#_bookmark10)], [Twidwell](#_bookmark37) [[2012](#_bookmark37)], [Twidwell et al.](#_bookmark40) [[2016](#_bookmark40)]). In the 2018 fire season, wildfire costs to the State of California alone were estimated at $148.5 (126.1–192.9) billion, or approximately 1.5 percent of the state’s annual $3 trillion GDP ([Wang et al.](#_bookmark44) [[2021](#_bookmark44)]). This figure includes direct costs, such as fire suppression costs and infrastructure destruction, and indirect costs such as human health impacts, lost livestock forage, and a depleted rural tax base. In the 2012 fire season, Nebraska wildfires burned more than 500,000 acres, destroying over 60 structures, and incurring fire suppression costs alone of nearly $12 million ([Nebraska](#_bookmark45) [Forest Service](#_bookmark45) [[2012](#_bookmark45)]). This single, direct cost (versus the myriad direct and indirect costs of California fires) equates to roughly 4 percent of the State of Nebraska’s entire 2012 budget ([Nebraska Legislature](#_bookmark46) [[2012](#_bookmark46)]). The U.S. Department of Commerce estimates the annual national cost of wildfires to be $71.1 billion to $347.8 billion.

Afforestation of U.S. Great Plains grasslands increases wildfire risk in a region without the historical precedent or infrastructure to manage such events ([Andrews and Rothermel](#_bookmark14) [[1982](#_bookmark14)], [Bidwell et al.](#_bookmark18) [[2008](#_bookmark18)], [Twidwell et al.](#_bookmark38) [[2013](#_bookmark38)], [Donovan et al.](#_bookmark7) [[2017](#_bookmark7)]), and presents material economic, ecological, and social risk to rural communities.

# The Potential Magnitude of Afforestation

A reanalysis of data from a study by [Nielsen et al.](#_bookmark8) ([[2014](#_bookmark8)]) indicates that an additional 61.2 million acres of rangeland, 12.8 million acres of pasture land, and 15.3 million acres of cropland could be afforested in the United States under a scenario in which carbon credits are priced at 50 USD metric ton-1 CO2-e. In this scenario, nearly half of the carbon sequestered from afforested lands (93.7 of the 200 M tons) comes from rangeland, i.e., previously grassland cover, acres (fig. 3), and much of this rangeland is concentrated in the U.S. Great Plains, where grasslands are already imperiled by conversion to cropland, fragmentation by urban and suburban sprawl, and suppression of reinforcing ecological processes, like fire, flood, and high intensity-short duration grazing.

# Conclusion

U.S. Great Plains grassland afforestation may initially appear to be a cost effective NCS pathway to generate the carbon offsets needed by governments, companies, and other entities to reach their decarbonization targets. Yet the social and ecological costs of grassland afforestation in the U.S. Great Plains are steep, and include increased vulnerability to wildfires, loss of grassland habitat and forage production, and the viability of long-term (10+ year) carbon storage on itself, especially if afforestation reduces soil carbon storage equilibria and wildfire occurs. Including such considerations in the scoping phase of an NCS project is therefore critical for carbon offset registries and project developers to align with the requirements of quality offset generation, which is to generate durable carbon offsets without incurring net harm for the people and ecology from whence those offsets are created.

# Funding

This work was funded by The Nebraska Game and Parks Commission, grant number xxx. The Nebraska Cooperative Fish and Wildlife Research Unit is jointly supported by a cooperative agreement between the US Geological Survey, the Nebraska Game and Parks Commission, the University of Nebraska-Lincoln, the US Fish and Wildlife Service, and the Wildlife Management Institute. The views set forth by contributors to this volume represent their own and do not represent the views of any public or private entity the contributor is affiliated with.

Diagram

Description automatically generated

**Figure 1:** Soil carbon (C) is a more stable, fire resistant form of ecosystem carbon storage relative to aboveground tree biomass, and its net storage capacity is determined by the type of plant carbon inputs (*Ix*) (left panel), and a soil’s intrinsic storage limits (i.e., when inputs and release in equilibrium) (right panel). As inputs increase, remaining storage capacity decreases. The greyed area in both panels shows gains in soil carbon when inputs increase. Intrinsic limits to carbon storage are understood to be the result of physico-chemical adsorption of C to clay surfaces ([Hassink](#_bookmark21) [and Whitmore](#_bookmark21) [[1997](#_bookmark21)]), physical protection of C in soil aggregates, the biochemical complexity or recalcitrance of the C inputs themselves, and other environmental limits on biological activity ([Baldock and Skjemstad](#_bookmark16) [[2000](#_bookmark16)], [Birge et al.](#_bookmark0) [[2015](#_bookmark0)]). Figures adapted from [Stewart et al.](#_bookmark36) ([[2007](#_bookmark36)]).

A picture containing diagram

Description automatically generated

**Figure 2:** Alternate scenarios of afforestation on soil carbon stocks over time with supporting literature and the predicted change in soil carbon storage equilibria according to the carbon saturation hypothesis. The scenarios include increased soil carbon (top; highlighted in green), no change in soil carbon (middle; highlighted in yellow) and decrease in soil carbon (bottom; highlighted in red) following grassland afforestation. Overall quantity and source of soil carbon (grassland vs forest) shifts subtly in each scenario, also according to the carbon saturation hypothesis ([Stewart et al.](#_bookmark36) [[2007](#_bookmark36)]).

Map

Description automatically generated

**Figure 3:** Rangeland acres vulnerable to afforestation when carbon is priced at $50 (C) per metric ton CO2. Reanalyzed from  [Nielsen et al.](#_bookmark8) [[2014](#_bookmark8)]

.

# References

P. L. Andrews and R. C. Rothermel. Charts for interpreting wildland fire be- havior characteristics. (September), 1982. doi: 10.2737/INT-GTR-131. URL <https://www.fs.usda.gov/treesearch/pubs/22647>.

D. G. Angeler, C. R. Allen, C. Barichievy, T. Eason, A. S. Garmestani,

N. a. J. Graham, D. Granholm, L. H. Gunderson, M. Knutson, K. L. Nash, R. J. Nelson, M. Nyström, T. L. Spanbauer, C. a. Stow, and S. M. Sundstrom. Management applications of discontinuity theory. *Jour- nal of Applied Ecology*, (Holling 1973):n/a–n/a, 2015. ISSN 00218901. doi: 10.1111/1365-2664.12494. URL [http://doi.wiley.com/10.1111/](http://doi.wiley.com/10.1111/1365-2664.12494) [1365-2664.12494](http://doi.wiley.com/10.1111/1365-2664.12494).

J. A. Baldock and J. O. Skjemstad. Role of the soil matrix and minerals in protecting natural organic materials against biological attack. *Organic* *Geochemistry*, 31:697–710, 2000.

N. N. Barger, S. R. Archer, J. L. Campbell, C. Y. Huang, J. A. Morton, and

A. K. Knapp. Woody plant proliferation in North American drylands: A synthesis of impacts on ecosystem carbon balance, 12 2011. ISSN 01480227.

T. G. Bidwell, D. M. Engle, M. E. Moseley, and R. E. Masters. Invasion of Oklahoma Rangelands and Forests by Eastern Redcedar and Ashe Juniper. *Oklahoma Cooperative Extension Service. Circular -947*, (April): 1–12, 2008. doi: 10.13140/RG.2.1.2330.0649.

H. E. Birge, R. T. Conant, R. F. Follett, M. L. Haddix, S. J. Morris, S. S. Snapp, M. D. Wallenstein, and E. a. Paul. Soil respiration is not limited by reductions in microbial biomass during long-term soil incubations. *Soil Biology and Biochemistry*, 81:304–310, 2015. ISSN 00380717. doi: 10.1016/j.soilbio.2014.11.028.

H. E. Birge, C. R. Allen, A. S. Garmestani, and K. L. Pope. Adap- tive management for ecosystem services. *Journal of Environmental Management*, pages 1–10, 2016a. ISSN 1098-6596. doi: 10.1017/ CBO9781107415324.004.

H. E. Birge, C. R. Allen, A. S. Garmestani, and K. L. Pope. Adaptive man- agement for ecosystem services. *Journal of Environmental Manage- ment*, 2016b. ISSN 1098-6596. doi: 10.1017/CBO9781107415324.004.

H. E. Birge, R. A. Bevans, C. R. Allen, D. G. Angeler, S. G. Baer, and D. H. Wall. Adaptive management for soil ecosystem services. *Journal of Environmental Management*, pages 6–13, 2016c. ISSN 1098-6596. doi: 10.1017/CBO9781107415324.004. URL [http://dx.doi.org/10.1016/j.](http://dx.doi.org/10.1016/j.jenvman.2016.06.024) [jenvman.2016.06.024](http://dx.doi.org/10.1016/j.jenvman.2016.06.024).

M. R. Busch and D. R. Presley. Cedar Afforestation of Prairie Alters Soil Properties on a Decadal Time Scale. *Soil Horizons*, 55(5):1–11, 2014. ISSN 2163-2812. doi: 10.2136/sh13-05-0015. URL [https://dl.](https://dl.sciencesocieties.org/publications/sh/abstracts/55/5/sh13-05-0015)

[sciencesocieties.org/publications/sh/abstracts/55/5/sh13-05-0015](https://dl.sciencesocieties.org/publications/sh/abstracts/55/5/sh13-05-0015).

F. S. Chapin. The Mineral Nutrition of Wild Plants. *Annual Review of Ecol- ogy and Systematics*, 11(1):233–260, 1980. ISSN 0066-4162. doi: 10. 1146/annurev.es.11.110180.001313. URL [http://www.annualreviews.](http://www.annualreviews.org/doi/10.1146/annurev.es.11.110180.001313) [org/doi/10.1146/annurev.es.11.110180.001313](http://www.annualreviews.org/doi/10.1146/annurev.es.11.110180.001313).

R. T. Conant. Sequestration through forestry and agriculture. *Wiley In- terdisciplinary Reviews: Climate Change*, 2(2):238–254, 2011. ISSN 17577799. doi: 10.1002/wcc.101.

V. Donovan, J. Burnett, C. Bielski, H. E. Birge, R. A. Bevans, D. Twidwell, and C. R. Allen. Social–ecological landscape patterns predict woody en- croachment from native tree plantings in a temperate grassland. *Ecology and Evolution*, 1(9), 2018. doi: 10.1002/eqe.3063.

V. M. Donovan, C. L. Wonkka, and D. Twidwell. Surging wildfire activity in a grassland biome. *Geophysical Research Letters*, 44(12):5986–5993, 2017. ISSN 19448007. doi: 10.1002/2017GL072901.

A. S. Elberg Nielsen, A. J. Plantinga, and R. J. Alig. Mitigating cli- mate change through afforestation: New cost estimates for the United States. *Resource and Energy Economics*, 36(1):83–98, 2014. ISSN 09287655. doi: 10.1016/j.reseneeco.2013.11.001. URL [http://dx.doi.](http://dx.doi.org/10.1016/j.reseneeco.2013.11.001) [org/10.1016/j.reseneeco.2013.11.001](http://dx.doi.org/10.1016/j.reseneeco.2013.11.001).

M. A. Finney, C. W. McHugh, I. C. Grenfell, K. L. Riley, and K. C. Short. A simulation of probabilistic wildfire risk components for the continental United States. *Stochastic Environmental Research and Risk Assessment*, 25(7):973–1000, 2011. ISSN 14363240. doi: 10.1007/ s00477-011-0462-z.

D. T. Fogarty, C. R. Allen, and D. Twidwell. Incipient woody plant en- croachment signals heightened vulnerability for an intact grassland re- gion. *Journal of Vegetation Science*, 33(6), 11 2022. ISSN 1100-9233. doi: 10.1111/jvs.13155.

E. J. Forster, J. R. Healey, C. Dymond, and D. Styles. Commercial afforesta- tion can deliver effective climate change mitigation under multiple de- carbonisation pathways. *Nature Communications*, 12(1), 12 2021. ISSN

20411723. doi: 10.1038/s41467-021-24084-x.

B. W. Griscom, J. Adams, P. W. Ellis, R. A. Houghton, G. Lomax, D. A. Miteva, W. H. Schlesinger, D. Shoch, J. V. Siikamäki, P. Smith, P. Wood- bury, C. Zganjar, A. Blackman, J. Campari, R. T. Conant, C. Delgado, P. Elias, T. Gopalakrishna, M. R. Hamsik, M. Herrero, J. Kiesecker, E. Landis, L. Laestadius, S. M. Leavitt, S. Minnemeyer, S. Polasky, P. Potapov, F. E. Putz, J. Sanderman, M. Silvius, E. Wollenberg, and J. Fargione. Natural climate solutions. *Proceedings of the National Academy of Sciences of the United States of America*, 114(44):11645– 11650, 2017

L. Guo and R. Gifford. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, 8:345–360, 2002.

M. E. Harmon. Addressing the Scale Question. *Journal of Forestry*, pages 24–29, 2001. ISSN 00221201.

J. Hassink and A. P. Whitmore. A Model of the Physical Protection of Organic Matter in Soils. *Soil Science Society of America* *Jour- nal*, 61(1):131, 1997. ISSN 0361-5995. doi: 10.2136/sssaj1997.

03615995006100010020x. URL [https://www.soils.org/publications/](https://www.soils.org/publications/sssaj/abstracts/61/1/SS0610010131) [sssaj/abstracts/61/1/SS0610010131](https://www.soils.org/publications/sssaj/abstracts/61/1/SS0610010131).

R. B. Jackson, J. L. Banner, E. G. Jobbágy, W. T. Pockman, and D. H. Wall. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature*, 418(6898):623–626, 2002. ISSN 1476-4687. doi: 10.1038/

nature00910. URL <https://doi.org/10.1038/nature00910>.

Y. Kishawi, A. R. Mittelstet, T. E. Gilmore, D. Twidwell, T. Roy, and

N. Shrestha. Impact of Eastern Redcedar encroachment on water re- sources in the Nebraska Sandhills. *Science of The Total Environment*, 858:159696, 2023. ISSN 0048-9697. doi: https://doi.org/10.1016/j. scitotenv.2022.159696. URL [https://www.sciencedirect.com/science/](https://www.sciencedirect.com/science/article/pii/S0048969722067961) [article/pii/S0048969722067961](https://www.sciencedirect.com/science/article/pii/S0048969722067961).

A. K. Knapp, S. L. Conard, and J. M. Blair. Determinants of soil CO2 flux from a Sub-humid grassland: effect of fire and fire history. *Ecological* *Applications*, 8(3):760–770, 1998.

Y. Kuzyakov and G. Domanski. Carbon input by plants into the soil. *Jour**nal* *of Plant Nutrition and Soil Science*, 163:421–231, 2000.

R. Lal. Forest soils and carbon sequestration. *Forest Ecology and Manage- ment*, 220(1-3):242–258, 2005. ISSN 03781127. doi: 10.1016/j.foreco.

2005.08.015.

D. C. McKinley, M. D. Norris, J. M. Blair, and L. C. Johnson. Altered Ecosystem Processes as a Consequence of Juniperus virginiana L. En- croachment into North American Tallgrass Prairie. pages 170–187. 2007. doi: 10.1007/978-0-387-34003-6{\\_}9.

D. C. Mckinley, M. G. Ryan, R. A. Birdsey, C. P. Giardina, M. E. Harmon, L. S. Heath, R. A. Houghton, R. Jackson, J. F. Morrison, B. C. Murray,

D. E. Pataki, and E. Skog, Kenneth. A synthesis of current knowledge on forests and carbon storage in the United States. *Ecological Applications*, 21(December 2010):1902–1924, 2011.

N. J. Mellor, J. Hellerich, R. Drijber, S. J. Morris, M. E. Stromberger, and

E. A. Paul. Changes in Ecosystem Carbon Following Afforestation of Native Sand Prairie. *Soil Science Society of America Journal*, 77(5): 1613, 2013. ISSN 0361-5995. doi: 10.2136/sssaj2012.0327. URL

<https://www.soils.org/publications/sssaj/abstracts/77/5/1613>.

S. L. Morford, B. W. Allred, D. Twidwell, M. O. Jones, J. D. Maestas,

C. P. Roberts, and D. E. Naugle. Herbaceous production lost to tree encroachment in United States rangelands. *Journal of Applied* *Ecology*, 2022. ISSN 13652664. doi: 10.1111/1365-2664.14288.

Nebraska Forest Service. Nebraska’s Wildfire Control Act: Collaborating to keep fires small. Technical report, 2012. URL [www.nfs.unl.edu](http://www.nfs.unl.edu/).

Nebraska Legislature. State of Nebraska FY2011-12 and FY2012-13 Bien- nial Budget. Technical report, 2012.

M. D. Norris, J. M. Blair, and L. C. Johnson. Land cover change in east- ern Kansas: litter dynamics of closed-canopy eastern redcedar forests in tallgrass prairie. *Canadian Journal of Botany*, 79(2):214–222, 2001a. ISSN 00084026. doi: 10.1139/cjb-79-2-214.

M. D. Norris, J. M. Blair, L. C. Johnson, and R. B. McKane. Assessing changes in biomass, productivity, and C and N stores following Junipe- rus virginiana forest expansion into tallgrass prairie. *Canadian Journal of Forest Research*, 31(11):1940–1946, 2001b. ISSN 0045-5067. doi: 10.1139/x01-132.

M. D. Norris, J. M. Blair, and L. C. Johnson. Altered Ecosystem Nitrogen Dynamics as a Consequence of Land Cover Change in Tallgrass Prairie. *The American Midland Naturalist*, 158(2):432–445, 2007. ISSN 0003-

0031. doi: 10.1674/0003-0031(2007)158[432:AENDAA]2.0.CO;2.

K. I. Paul, P. J. Polglase, P. K. Khanna, J. G. Nyakuengama, A. M. O’Connell, T. S. G. Battaglia, and M. Battaglia. Change in Soil Carbon Following Afforestation or Reforestation. *Forest Ecology and Manage-* *ment*, 168:241–257, 2001.

A. J. Plantinga and J. Wu. Co-Benefits from Carbon Sequestration in Forests: Evaluating Reductions in Agricultural Externalities from an Af- forestation Policy in Wisconsin. *Land Economics*, 79(1):74–85, 2006. ISSN 0023-7639. doi: 10.2307/3147106.

W. M. Post and K. C. Kwon. Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology*, 6(3):317–327, 3 2000. ISSN 1354-1013. doi: 10.1046/j.1365-2486.2000.00308.x. URL <http://doi.wiley.com/10.1046/j.1365-2486.2000.00308.x>.

A. J. Reisinger, J. M. Blair, C. W. Rice, and W. K. Dodds. Woody Vegetation Removal Stimulates Riparian and Benthic Denitrification in Tallgrass Prairie. *Ecosystems (New York)*, 16(4):547–560, 2013. ISSN 1432-9840. doi: 10.1007/s10021-012-9630-3.

P. M. Rice, G. McPherson, and L. Rew. Fire and nonnative invasive plants in the interior west bioregion. *Wildland Fire in Ecosystems Fire and Nonnative Invasive Plants*, 6(September):141–171, 2008.

J. S. Russell. Soil fertility changes in the long-term experimental plots at kybybolite, south australia ii. Changes in phosphorus. *Australian Jour- nal of Agricultural Research*, 11(6):926–947, 1960. ISSN 00049409. doi: 10.1071/AR9600926.

F. Samson and F. Knopf. Prairie conservation in North America. *Bio- Science*, 44(6):418–421, 6 1994. ISSN 0006-3568. doi: 10.2307/

1312365. URL [https://academic.oup.com/bioscience/article/44/6/](https://academic.oup.com/bioscience/article/44/6/418/255658) [418/255658](https://academic.oup.com/bioscience/article/44/6/418/255658).

W. H. Schlesinger. Carbon Balance in Terrestrial Detritus. *Annual Review of Ecology and Systematics*, 8:51–81, 1977.

J. O. Skjemstad, P. Clarke, J. A. Taylor, J. M. Oades, and S. G. McClure A. The chemistry and nature of protected carbon in soil. *Australian Journal of Soil Research*, 34(2):251–271, 1996. ISSN 00049573. doi: 10.1071/ SR9960251.

C. E. Stewart, K. Paustian, R. T. Conant, A. F. Plante, and J. Six. Soil carbon saturation: Concept, evidence and evaluation. *Biogeochemistry*, 86(1): 19–31, 2007. ISSN 01682563. doi: 10.1007/s10533-007-9140-0.

D. Twidwell. *FROM THEORY TO APPLICATION: EXTREME FIRE, RESILIENCE, RESTORATION, AND EDUCATION IN SOCIAL- ECOLOGICAL DISCIPLINES*. PhD thesis, 2012.

D. Twidwell, B. W. Allred, and S. D. Fuhlendorf. National-scale assessment of ecological content in the world’s largest land management framework. *Ecosphere*, 4(8):1–27, 2013.

D. Twidwell, W. E. Rogers, C. L. Wonkka, C. A. Taylor, and U. P. Kreuter. Extreme prescribed fire during drought reduces survival and density of woody resprouters. *Journal of Applied Ecology*, 53(5):1585–1596, 2016. ISSN 13652664. doi: 10.1111/1365-2664.12674.

U.S. Bureau of Economic Analysis (BEA). GDP by State, 2022.

P. Vitousek. *Nutrient Cycling and Limitation*. Princeton University Press, 2004. ISBN null. doi: 10.2307/j.ctv39x77c. URL [http://www.jstor.](http://www.jstor.org/stable/j.ctv39x77c) [org/stable/j.ctv39x77c](http://www.jstor.org/stable/j.ctv39x77c).

D. Wang, D. Guan, S. Zhu, M. M. Kinnon, G. Geng, Q. Zhang, H. Zheng,

T. Lei, S. Shao, P. Gong, and S. J. Davis. Economic footprint of Cali- fornia wildfires in 2018. *Nature Sustainability*, 4(3):252–260, 3 2021. ISSN 23989629. doi: 10.1038/s41893-020-00646-7.

B. P. Wilcox, S. D. Fuhlendorf, J. W. Walker, D. Twidwell, X. B. Wu,

L. E. Goodman, M. Treadwell, and A. Birt. Saving imperiled grass- land biomes by recoupling fire and grazing: a case study from the Great Plains. *Frontiers in Ecology and the Environment*, 20(3):179–186, 4 2022. ISSN 15409309. doi: 10.1002/fee.2448.