The spatial and temporal domains of natural climate solutions

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This paper is a non-peer reviewed preprint submitted to EarthArXiv.

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Abstract

Natural climate solutions (NCS) have been proposed to mitigate climate change by removing CO_2 from the atmosphere and increasing organic carbon storage in ecosystems. Adoption is required at global scales, but implementation of NCS have been limited by the lack of a systematic framework to prioritize ecosystem restoration or conservation at local and regional scales. Current carbon sequestration policies at the national scale often fail to consider local and regional ecological feedback systems and tradeoffs among finite natural resources. These have unintended effects on the carbon permanence of ecosystems, defined as the residence time of carbon (C) before release to the atmosphere as CO_2. By combining estimates of soil organic C stocks, land use, projected precipitation changes, and landscape-level analysis of carbon and water flux in Oregon and Washington, we show that NCS efforts should be prioritized in natural areas with low soil C stocks and projected future precipitation increases. On the other hand, conservation may be more appropriate for regions with high soil C stocks and projected decreases in precipitation. Our consideration of geography acknowledges the ecological and socioeconomic challenges to NCS implementation and allows for the identification of high-priority sites for NCS. This protocol can be adapted at local and regional scales to guide policy for targeting the highest-priority locations for implementation of NCS.

1. Introduction

Natural climate solutions (NCS) consist of land conservation, restoration, and management efforts that have potential to mitigate climate change by removing CO_2 from the atmosphere into terrestrial ecosystems (e.g., Griscom et al., 2017). Examples of NCS practices include reforestation and other forest management practices, regenerative agricultural practices including rotational grazing, carbon farming and biochar application, and wetland restoration (Graves et al., 2017). At the global scale, NCS seem to be readily deployable techniques that can contribute to sequestration of atmospheric CO_2 (Fargione et al., 2018). However, a global drawdown project requires feasible processes to promote action on local and regional levels (Bossio et al., 2020). Specifically, priorities must be regional and acknowledge social and ecological factors within and across different landscapes. The central question to be answered in this work is: how can we prioritize local and regional action towards NCS?

Successfully implementing NCS requires a framework that is congruent at multiple spatial and temporal scales. Furthermore, this framework must also consider bioclimatic factors that can be camouflaged within large-scale objectives. Not all landscapes may be appropriate for NCS projects (e.g., competing socioeconomic uses, or too dry to plant trees) and thus no single NCS method is appropriate for all bioclimatic conditions (Baldocci et al., 2018). We propose a geographic-based framework for guiding regional prioritization of NCS implementation using soil carbon stocks (a proxy for existing sequestration capacity) and projected precipitation changes (a control on future sequestration capacity) across ecoregions. In this study, the ecoregions of an area are considered in terms of geographic extent, baseline carbon conditions, and climate. In doing so, the broad-scale and sector-specific NCS can be ranked, prioritized, and tailored to locally-relevant practices. Within the identified NCS priority areas, however, we caution that implementation measures must utilize ecologically appropriate species, and that overall sequestration capacity is limited by current bioclimatic conditions.

We chose Oregon and Washington to serve as examples of ecoregions due to the myriad biological, geological, and climatologic features that exist within Pacific Northwest ecosystems. Adopting Peter Vitousek's theory arising from consideration of the Hawaiian Islands, we argue that the region can act as a laboratory within the field of NCS where "fundamental mechanisms can be identified, understood, and tested" (Vitousek et al., 1987).

2. Methods

We first gathered spatio-temporal data from previously published works on natural climate solutions to identify a general distribution of spatial and temporal scales that have been considered in the literature **(Figure 1).** We reviewed 30 studies published from 2011 – 2021 that contained the keywords "Natural Climate Solutions" or "Nature-based Solutions". Approximately 26 of the 30 studies used a modelling approach for estimating CO_2 sequestration potential of NCS, which allowed for approximating the spatial and temporal extent considered in each study (**Supplementary Data**). The frequencies of spatial and temporal observations are displayed as histograms to illustrate the general scales that have been traditionally considered in theoretical and experimental approaches to NCS implementation **(Figure 1**). If no specific spatial scale was mentioned by the original study authors, we approximated the spatial and temporal scales of the study by a) estimating the spatial extent of the modelled study area as categorical data (e.g., global, national, regional, or local) which were then binned into the approximate area of proposed NCS implementation (km^2). Temporal scales of published studies, if not author-identified, were approximated by using boundary conditions of the study. For example, if a study estimated the climate mitigation potential in 2020 and 2100 ($GtCO_{2e}$ yr^{-1}) of a NCS, we approximated the temporal extent of the study to be 20-100 years.

Next, we used widely available geospatial data that caters to local and/or regional scales to prioritize areas for NCS implementation, ensuring a replicable approach **(Figure 2).** Bioclimatic regions are defined here by Washington and Oregon state Level III ecoregions, originating from the EPA as "areas of general similarity in ecosystems and in the type of quality, and quantity of environmental resources". These regions are necessary for the proper structure and execution of ecosystem management strategies across agencies and non-governmental organizations accountable for the natural resources within a geographic area (EPA, 2020). Terrestrial baseline soil carbon was based off 250 m spatial resolution soil data from the World Soil Information Service database and accessed via SoilGrids™ version 2.0,.Tidal wetlands carbon was derived from the North American Carbon Project (soilgrids.com, Hinson et al. 2019). Land use data was categorized as developed versus natural lands and was derived from the Washington State Department of Ecology 2010 State Land use dataset, and the Oregon Development Project – 2014 dataset, originated by the USFS Pacific Northwest Research Station. The projected precipitation data was collected via the Downscaled CMIP3 and CMIP5 Climate and Hydrology Projections (https://gdo-dcp.ucllnl.org/downscaled_cmip_projections/), which is a collaborative archive containing fine spatial resolution climate projections over the contiguous United States. The CMIP5 multi-model ensemble output utilizes 1/8° resolution bias corrected precipitation projections that were produced by running period analyses using the MIROC5 (Model for Interdisciplinary Research on Climate) global climate model and either RCP2.6 as an emission scenario that leads to low greenhouse gas concentrations, or RCP8.5, which is a baseline scenario in the absence of a climate change policy. RCP2.6 assumes substantial changes in energy use, but a large increase in global cropland area This scenario was chosen based upon these traits, while RCP8.5 was chosen to represent the business-as-usual scenario. MIROC5 is an atmosphere-ocean general circulation model that was selected for its wide-use with improved results especially found in precipitation (Watanabe et al., 2010), created by the Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology.

3. Results and Discission

3.1 Observational scales of Natural Climate Solutions

A compilation of observational scales from previously published literature shows that spatial and temporal approaches to examining NCS have two broad distributions (Figure 1). First, the spatial extent of NCS research has largely been focused at the regional, national and global scales. For example, many studies that take a modelling approach to assessing NCS feasibility are focused on large (national and global) scales. The temporal scale of NCS implementation, e.g., the duration of carbon permanence, has a normal distribution, with most studies considering temporal scales of 20-100 years (Figure 1) for carbon permanence. A NCS focus on such large spatial scales and may fail to consider local and regional challenges to NCS. Similarly, the focus of carbon permanence on short temporal scales (e.g., 20 years) may limit the long-term sustainability of NCS implementation.

It should be noted that our results for approximating scales of NCS were potentially influenced by several method-based issues which may have potential biases and uncertainty in quantification of scales. Several studies that modelled C sequestration potential of NCS did not precisely state the observational scales used, and therefore we had to estimate the scale values for most observations, especially those focused on the national or global scale. Estimation errors may have therefore biased our findings. In any case, this compilation provides a general reference frame to interpret the most common observational scales currently used in the consideration of NCS implementation.

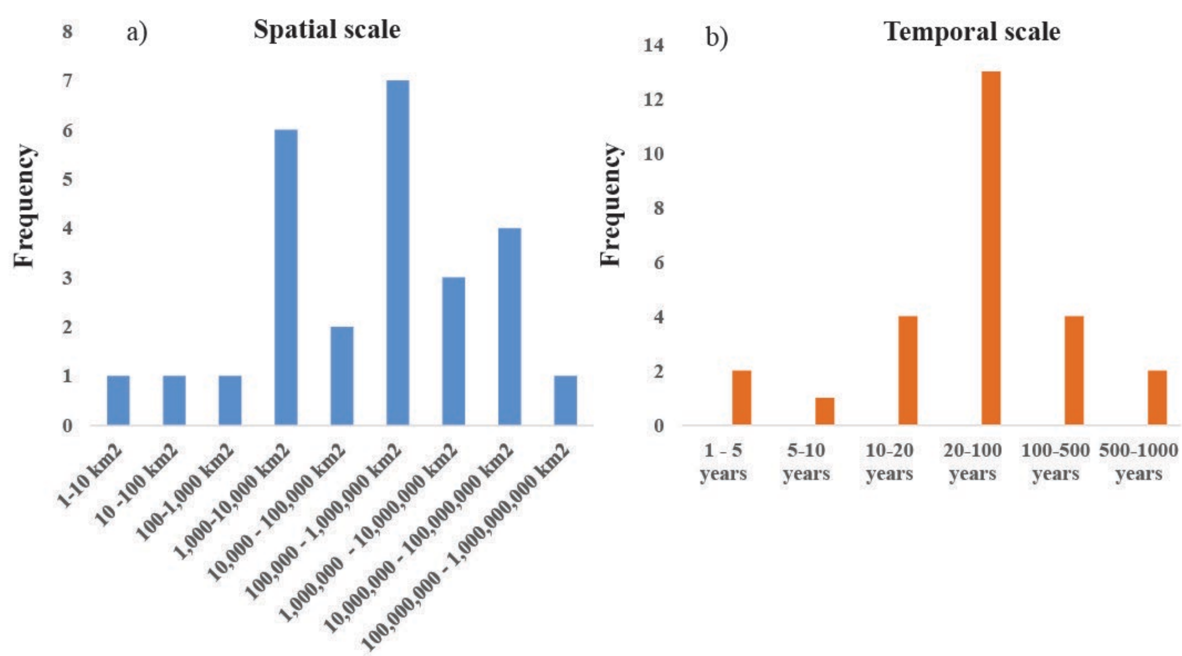

Figure 1. Observational data from previously published work showing the general spatial and temporal domains of natural climate solutions. a-b, Histograms of a) the spatial scale of natural climate solution (NCS) implementation and b) temporal scales of carbon permanence collectively gathered from a survey of 30 studies focused on natural climate solutions

3.2 Guiding Tradeoffs for NCS Prioritization

Our geographical framework for NCS implementation is explicitly focused on two challenges within the implementation process: (1) delineating locations that present opportunities for carbon drawdown, and (2) identifying areas that would benefit more from conservation than

NCS implementation (Figure 2). Our spatial approach to developing NCS strategies is guided by the tradeoffs that are inherent in ecosystem mining of CO_2 from the atmosphere (Baldocci and Penuelas, 2018). We use the carbon-water relationship as an example of a tradeoff that applies across locations and spatial scales and can be leveraged to describe the site-specific implications of NCS (Figure 3).

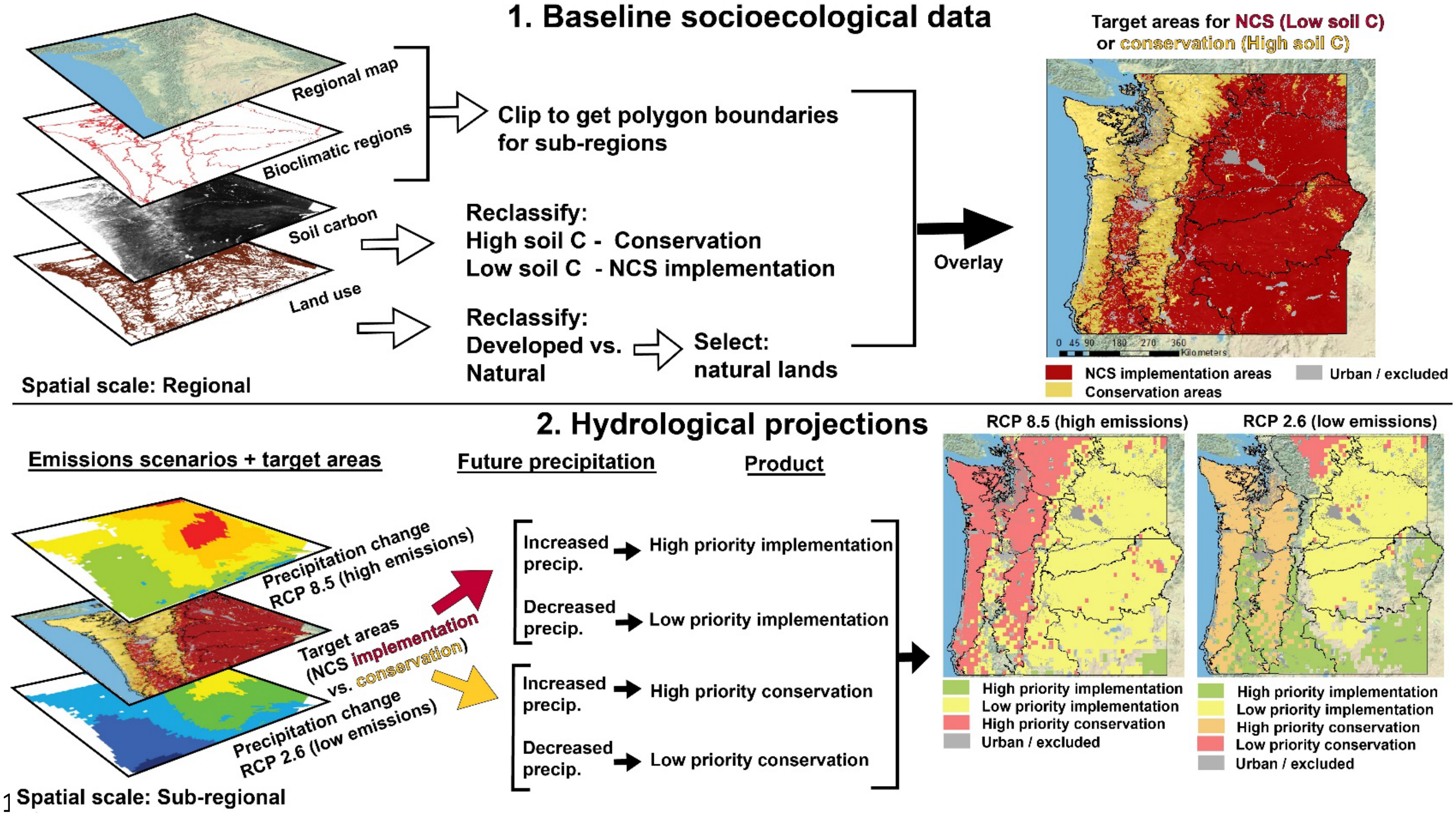

Figure 2. Data-driven framework for prioritization of natural climate solutions (NCS) implementation areas or conservation across regional and local spatial scales. Step 1 uses baseline data of a socio-ecological context including bioclimatic zones, soil carbon stocks, and land use to identify target areas for NCS implementation and conservation, respectively. Step 2 combines those target areas with future projections of precipitation using both high and low emission scenarios to rank areas by priority. In Step 1, Soil C was reclassified as high if C stocks were C stocks <3

The assimilation of carbon through vegetation is physically and biologically tied to water use and plant morphology (**Figure 3).** Gross primary production (GPP) scales with evapotranspiration (ET) and Leaf Area Index (LAI) (**inset, Figure** 3). The linear regression shows that on average evapotranspiration increases by 0.247 millimeters for each increase in GPP per gram of carbon per square meter in a year ($ET = 213.4 + 0.247(GPP)$). This is consistent with previous studies comparing annual estimates of CO_2 and water vapor exchange (Law et al., 2002). The scaling of GPP with LAI suggests that ecosystem functions relevant to NCS are strongly tied to plant morphology. The role of plant morphology in defining ecosystem function can be extended to water use. For example, the vein density of leaves can be used to predict leaf conductance to water vapor (Boyce et al., 2009).

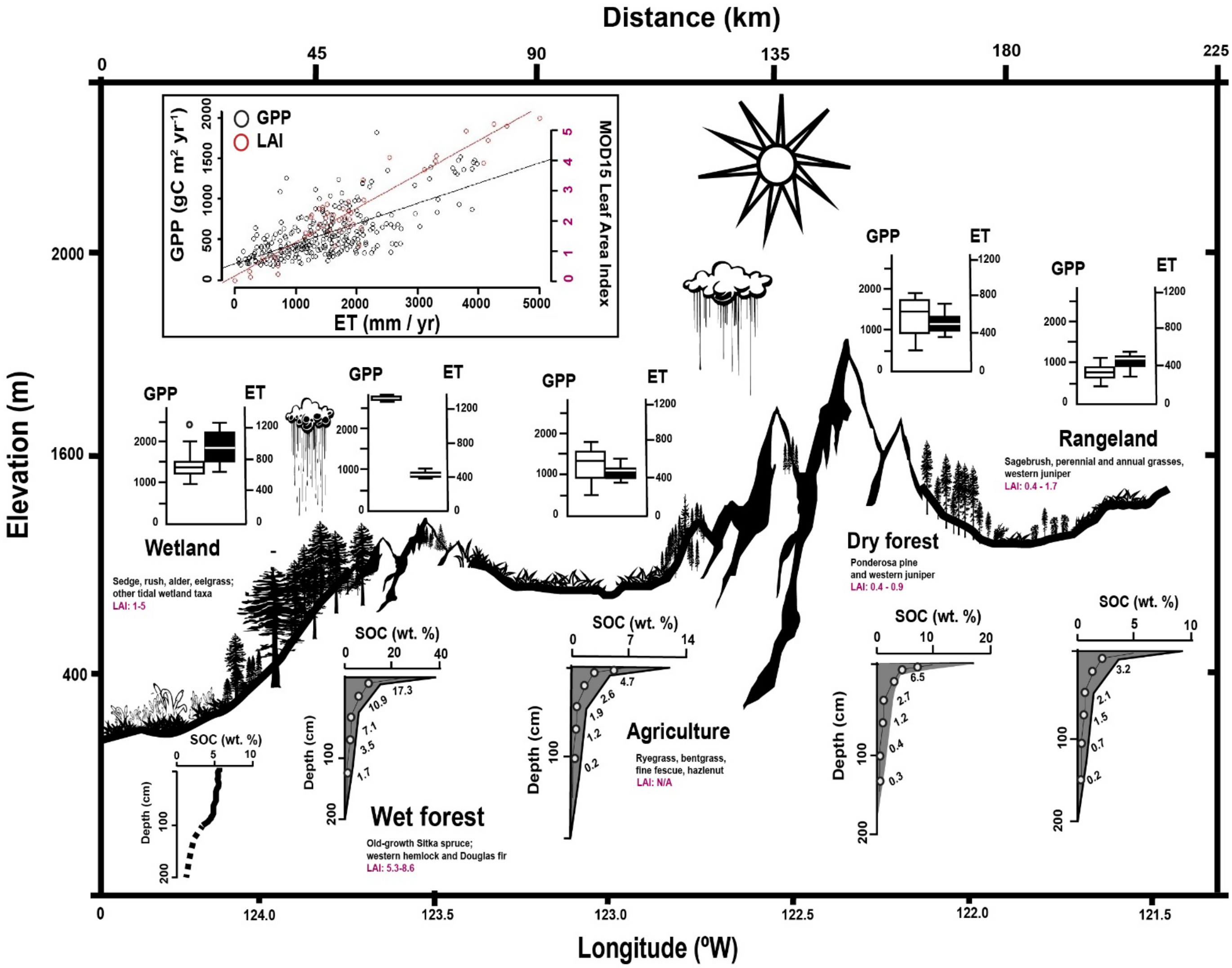

Figure 3. Carbon-water tradeoffs across a hypothetical Pacific Northwest transect. Inset figure shows relationships between evapotranspiration (ET), gross primary productivity (GPP), and leaf area index (LAI). Data for the GPP and ET relationship were adapted from Baldocchi and Penuelas (2018) who used the R package 'digitize'. The GPP and ET values represent annual averages of eddy covariance measurements from the FLUXNET 2015 dataset that includes 155 global flux towers (Supplemental Table 1). The resulting linear regression is $ET = 213.4 + 0.247(GPP)$. Data for the LAI and ET relationship are from Hashimoto et al. (2012). LAI values represent annual averages from 2001-2008 from MODIS product cutouts surrounding 21 FLUXNET towers (**Table S2).** Represented by the equation $LAI = 0.76 + 0.00153(ET)$. Boxplots show the carbon-water tradeoff (GPP and ET) across ecosystems (data from Baldocci and Penuelas, 2018 and **Kwon et al., 2018 [wet forest data]).** Modelled soil organic carbon (SOC) data were derived from SoilGrids™ (soilgrids.org) by randomly selecting a point within each ecosystem (except wetlands) across a transect from Cape Perpetua, OR to Burns, OR. Mean derived SOC values are listed next to each data point. Modelled mean SOC from soilgrids.org was computed using a default random forests algorithm in the R package 'ranger'. The 0.05 and 0.95 quantiles (gray shaded area) present the lower and upper boundaries of a 90% prediction interval and are used as a measure of prediction uncertainty. Quantiles of the

distribution were computed with Quantile Regression Forests. Representative wetland SOC is after Hinson et al. (2017) using National Wetlands Inventory data and the U.S. Soil Survey Geographic Database to model the area-weighted wetland SOC to 100 cm depth across all U.S. west coast tidal wetlands; SOC is estimated and not modelled from 100-200 cm (dotted line, wetland SOC).

The significant variability about the mean regression line (**Figure 3, inset**) is an important indicator of differences in ecosystem scale water use efficiency towards carbon drawdown. A reasonable interpretation of this variability is that points falling below the regression line are more water efficient than points falling above the regression line. Species composition is one of the primary sources of variation in the relationship between GPP and ET, and the regression between LAI and GPP in part explains the role of species composition in assimilating CO_2. However, this relationship does not fully explain the variability in the ET – GPP data. Characterizing the sources of this variability can support efforts to optimize the water – carbon balance through NCS implementation. Successful NCS implementation will require optimizing CO_2 fixation per unit of water lost to evapotranspiration, which relies upon characterizing the sources of variation in the water – carbon balance.

For example, within the Pacific Northwest, two primary forest types occur. Mesic forests throughout the western Cascade and Coast Mountain Ranges are typically dominated by Douglas-fir (*Pseudotsuga menziesii*), while dry forests dominated by ponderosa pine (*Pinus ponderosa*) cover the eastern Cascades and interior mountain ranges (Figure 2). Physiological differences cause inter- and intraspecific variation in growth rates throughout these forests. In general, trees growing in mesic conditions have higher GPP and lower WUE (water-use efficiency) than their more arid counterparts (Ruzica et al. 2017). Driven by biotic and abiotic conditions, water use efficiency varies across species, space, and time (Silva and Lambers, 2020). As a result, these forests have higher carbon sequestration capacities but also higher rates of water loss through transpiration. In contrast, trees growing in more arid conditions have less of a water cost for carbon assimilation. The water-carbon tradeoff is of significant consequence to the applicability of NCS implementation at a regional-scale. In Oregon, the GPP of a mesic Douglas fir-dominated wet forest is 2590 $gCm^{-2}yr^{-1}$ versus 1631 $gCm^{-2}yr^{-1}$ in dry forests of ponderosa pine approximately 160 km to the east (Kwon et al. 2018). Critically, the intrinsic water use efficiency is approximately four times greater in the semi-arid ponderosa pine forest (Kwon et al. 2018). Tree ring analysis of Douglas-fir sites along a 160 km North-South transect in western Oregon show greater growth in wetter sites versus higher intrinsic WUE in drier sites (Ruzica et al. 2017). While moisture availability is primary control on plant growth, additional parameters such as nutrient availability, growing season length further constrain NCS potential. We have shown areas where NCS implementation is desirable, but there are additional considerations that we believe will ultimately determine the effectiveness of NCS, such as current land cover type, which has implications for increasing carbon permanence across natural and managed landscapes. Prioritization of NCS across several natural ecosystems of the Pacific Northwest “laboratory” are discussed in the following paragraphs.

3.3 Wet Forest

Some of the Earth's carbon-rich forests lay within the temperate zone in the humid Pacific Northwest. These late successional old-growth forests have stockpiled carbon in both biomass and soil. The measured carbon densities from these forests are higher than all other types of vegetation across the globe (Smithwick et al., 2002). However, with warming, these forests could become carbon sources, as increasing temperatures impact plant and soil respiration rates and the inter-annual variability of net ecosystem exchanges (Baldocchi et al., 2018). Climate projections suggest an increased warming and increased precipitation for the region, where droughts are likely to occur between intense precipitation storms (IPCC, 2014).

The projected warming and fluctuations in wetness for this region will likely impact productivity rates as the photosynthetic response of mid- and high-latitude plants to increased warming depends upon soil moisture, where the effect will be positive when moisture is sufficient and negative were soils are drier (Reich et al., 2018). This is one example of the trade-offs that occur between water and carbon that have implications for this ecosystem to sequester carbon. **Figure 3** focuses on coastal forests as an example of wet forests, where the boxplots of GPP and ET indicate that the range of each is similar, and the cost of this level of GPP requires more ET in comparison to other ecosystems. These trade-offs are not ubiquitous for all wet forests, however, in fact they are dependent upon management history and microclimate. Recent research suggests that intensive management practices in the western cascades of Oregon can decrease water-use efficiency in Douglas-Fir trees, thereby increasing their vulnerability to droughts (Creutzberg et al., 2017). The coastal forest eco-region in the PNW has been more intensely harvested in comparison to the cascades due to its largely fragmented patchwork of ownership with differing management goals (Creutzburg et al., 2017). The climate of the two eco-regions can differ considerably from the interaction of maritime and continental air masses that impacts precipitation amounts, timing and whether it falls as snow or rain (Waring & Franklin, 1979). NCS that consider these geographic constraints are more likely to be successful, as conserving ecoregions with high levels of baseline carbon stocks will be dependent upon these factors impacting water-carbon tradeoffs.

With the goal of increasing carbon permanence, our framework suggests NCS in this region should be focused on conserving existing carbon in the form of woody biomass and soil organic carbon. Ecological Forestry, a more sustainable management practice, has been endorsed by creators of the Northwest Forest Plan (NWFP), which substantially reduced harvesting of old-growth forests, yet still promotes controversial practices such as clearcutting and large-scale thinning. Ecological forestry attempts to replicate early seral ecosystems where instead of clearcutting an entire stand of trees, the removal of smaller amounts of trees and longer harvest cycles promote carbon sequestration and overall forest health relative to clearcutting (Franklin et al., 2018). According to Forests Under Pressure, a recent analysis of local responses to global issues, place-based collaborative groups are more likely to succeed in providing sustainable management plans considering broader viewpoints as opposed to expert-only decision-making (Moseley & Winkel, 2014). These collaborative organizations offer a venue for getting beyond cultural and socio-economic barriers that would prevent a paradigm-shift away from profit-based management, towards a more ecological-social-economic framework for land-use and NCS implementation.

3.4 Dry Forest

The dry forests of the eastern Cascades, Klamath, and interior mountains are dominated by ponderosa pine and a frequent, low to mixed-severity fire regime (Halofsky et al., 2020). In this region, projected changes include increasing temperatures and hydrologic cycle intensification resulting in increased drought frequency and severity. These changes are anticipated as moisture loss during the prolonged summer dry season is amplified by warmer temperatures while precipitation increases are restricted to winter months. Previous research has shown increases in the annual number of large fires in the western United States that were consistent with increased drought severity and reflect long-term patterns that are to be expected with climatic changes (Dennison et al., 2014). Both plant species composition and ecosystem productivity are projected to occur due to changes in growing-season length, precipitation, and climatic water deficits (Westerling, 2016). As a result, fire regimes are expected to be impacted by interacting effects of short-term variations in productivity and long-term vegetation shifts. These shifts will not only impact NCS capacity, but also the potential permanence of carbon sequestered.

Fuel is the most significant control on fire severity, followed by fire weather, climate, and topography, respectively (Parks et al. 2018). As a consequence, land management decisions have critical implications for amount of carbon released during a fire event as well as the trajectory of ecosystems post-fire. This is especially important given evidence that the frequency of extreme fire weather is increasing (Parks et al. 2018; Davis et al. 2019). Additionally, patterns of repeat fires suggest that fine fuel accumulation is a strong control on disturbance frequency intervals. This pattern is spatially dependent (coastal areas with higher productivity can re-burn sooner than drier interior areas), indicating that the structure of forest vegetation is an important component of fire regime frequency and severity (Buma et al., 2020). As a result, all NCS activities must take in to account the preexisting fire regime to ensure carbon permanence. Planting trees within a forest that is primed to burn may only serve to increase fuel loading and potential fire severity.

Managed burning (the alternative to active fire suppression) under non-extreme fire weather conditions has been suggested as a pathway to decrease fire severity over the next century (Parks et al., 2016). A small proportion of fires are responsible for ~95% of the annual area burned. By aggressively suppressing low-severity fires, we paradoxically select for high-severity fires by increasing fuel on the landscape (Dunn et al., 2017). As a result, when fires do escape (because of high fuel loads and extreme fire weather), they burn at higher severity and release more carbon (Parks et al., 2018). The combination of high fuel loads resulting from a century of fire suppression activities, warmer climatic conditions and extreme fire weather interact to increase the frequency of high-severity, stand-killing fires (Taylor et al., 2016). Ultimately, NCS activities must integrate fuels and fire management planning into their implementation in order to be successful. Current bioclimatic conditions and modeled changes in precipitation serve as a pathway to understanding the predominant fire regime and help to inform appropriate landscape-specific NCS actions (**Figure 2**).

3.5 Tidal Wetlands

Tidal wetlands are valuable carbon sinks present within the Coast Range ecoregion. The wetlands have been reduced for agriculture and pastureland via implementation of dikes (Boule and Bierly, 1987), yet legislative protections in the Pacific Northwest have limited conversions, and therefore the potential for reducing GHGs is estimated to be limited to ~5200 ha of degraded area (Graves et al., 2020). Despite the limited geographic extent, wetlands hold the highest carbon per unit area (Griscom et al., 2017) and the Pacific Northwest coast contains some 1.4-1.6 million tons of carbon (Hinson et al., 2019). Albeit less influenced by precipitation changes than other ecosystems, net CO_2 emissions from restored tidal wetland is dependent on initial greenhouse gas inputs, rainfall, and elevation (Negandhi et al., 2019). Tidal reinstatement in higher elevation sites can result in a lower CO_2 emissions and unchanged CH_4 emissions when large rain events occur (Negandhi et al., 2019).

The water-carbon tradeoff for NCS implementation in tidal wetlands is distinct from other ecosystems considered in this work. For example, high ET may not be as detrimental to this aquatic landscape as it is to other ecosystems in our model region (**Figure 3**). Instead, land management actions such as dredging, nutrient loading in the watershed via logging or runoff, coastal development, and introduction of invasive species are more serious threats to tidal wetlands and seagrass establishment with the potential to convert even reinstated tidal wetlands from carbon sinks to sources (Short and Wyllie-Echeverria, 1996). This “coastal squeeze” in which coastal ecosystems like seagrass meadows and salt marshes, both of which enhance carbon storage through aquatic vegetation, are stuck between sea level rise, increasing sea temperatures, and coastal infrastructure, is a prominent threat to tidal wetlands (Mills et al., 2016).

21st century advancements in monitoring methods for blue carbon (BC) systems, such as seagrass meadows, salt marshes, and mangroves, have allowed for more frequent monitoring at larger spatial scales. Remote sensing approaches are now common for estimating seagrass coverage and density, including side scan sonar, passive spectral sensors used on boats and kayaks (Nahirnick, 2019), manned and unmanned aircraft (Duffy et al., 2018), and satellites. Higher resolution aerial imagery has been the standard data format for mapping seagrass, as more complex, fragmented seascapes are delineable and timing of image acquisition is more flexible (Costello and Kenworthy 2011, Uhrin and Turner, 2018). Furthermore, the usage of unmanned aerial vehicles (UAVs), commonly known as drones, are becoming increasingly preferable among conservation groups due to their relative low cost, flexibility of flight planning, and ability to capture seagrass health indicators at fine scales (Duffy et al., 2018, Ventura et al. 2016). Recent research has suggested that complex and fragmented patch shapes are more susceptible to negative effects of poor environmental conditions and could indicate critical transitions in the landscape state (Carr et al., 2010, Uhrin and Turner, 2018). Patch dynamics related to this phenomenon, such as the number of patches, patch shape, and leaf area index have been monitored using high resolution satellites, multispectral and hyperspectral airborne sensors, and UAV technology (Fyfe 2003, Lyons et al., 2015, O'Neill et al., 2013, Frederiksen et al., 2004, Barrell and Grant, 2015, Ventura et al., 2016, Duffy et al., 2018). These metrics provide insight to within-patch heterogeneity that can reveal fine-scale signs of degradation or recovery previously unattainable by traditional monitoring methods. A recent study found that over 120 studies on remote sensing for BC ecosystems monitoring have been conducted since 2010 (Pham et al. 2019). Application of these and other monitoring method in NCS can serve as catalyst for increasing C storage in coastal wetlands.

Varying soil organic carbon stock in PNW estuaries also provides opportunities and challenges for prioritizing NCS implementation. In general, carbon storage in estuaries varies by latitude and estuary size; smaller estuaries may have < 0 T C stored (Hinson et al., 2019). Smaller tidal wetland zones are more sensitive to dredging, the "coastal squeeze", and poor water quality. Seagrass presence can increase carbon storage and can delineate where conservation vs. implementation should take place within these "blue carbon" zones. In other words, tidal wetlands with dense seagrass meadows should be preserved, whereas areas that are bare but capable of hosting native species, such as eelgrass (*Zostera marina)*, should be targeted for restoration. For example, eelgrass restoration typically focuses on total cover, total area, and shoot density. Local hydrodynamics, sediment composition, predators, and navigability impact which restoration technique is best, but the two most common styles are transplantation from donor beds or seed broadcasting. Monitoring is commonly done seasonally for two years or more, where vegetation characteristics, biotic, and abiotic factors are measured, of which can be utilized within other NCS objectives that involve monitoring of environmental quality conditions. Seagrass Net, an international monitoring program for seagrasses, provides standard protocol for monitoring seagrasses using quarterly monitoring of permanent plots to detect significant change within 1-2 years (Mckenzie et al., 2003). As seagrass restoration has low success rates in the Pacific Northwest (Stamey, 2004, Fonseca et al., 1988), removing dikes to reinstate tidal wetlands can further increase the carbon sink and encourage seagrass expansion and eradication of invasive species (Herrick and Wolf 2005). Restoration versus conservation of tidal wetlands has implications for net carbon storage, but equally as important is the consideration of the temporal scales of carbon permanence in this ecosystem and others. How long will sequestered carbon be stored? Overall, efforts to prioritize NCS implementation at local and regional scales should consider the temporal scale of carbon permanence across areas containing both terrestrial and blue carbon.

3.6 Carbon Permanence

The success of NCS fundamentally depends on carbon permanence, defined as the mean residence time of sequestered carbon across landscapes. Across tidal wetlands, wet forests, rangelands and dry forests, how long will carbon (C) be sequestered after NCS implementation?

What use are NCS if sequestered C is rapidly released? We use carbon permanence here to describe C sequestration over timescales relevant to the current carbon budget outlined by Goldstein et al. (2018) to limit warming to 1.5° C above preindustrial levels. Natural climate solutions have the potential to sequester between 100-550 Gt CO_2 between 2018 and 2100 at a cost of less than $100 per ton carbon (Griscom et al., 2017), but ensuring that sequestered C remains "permanent" determines the ultimate success of NCS. For example, simple approaches to afforestation or reforestation which may consider soil type (Bastin et al; 2019) do not consider baseline soil organic carbon stocks, which if altered through uniform NCS across landscapes could result in unintentional liberation of C from areas with high baseline soil C and/or biomass C (e.g., wetlands or old-growth forests). We argue these areas should be conserved and not subject to NCS (Figure 1). Inherent differences in soil C stocks across ecosystems demand regional and sub-regional solutions for encouraging long-term soil C sequestration.

On land, soil organic matter (SOM) is the largest terrestrial pool of organic C, containing more organic C than global vegetation and the atmosphere combined (Lehmann and Kleber, 2015). Spatial prioritization of NCS requires a baseline assessment of SOM: How old is soil C, and how much variation is there on C residence time across ecosystems? A multitude of environmental and biological controls determine C stability in soils (Schmidt et al., 2011) and sediments (Nelson and Baldock, 2005). Sorption of organic molecules to the surfaces of minerals and amorphous colloids influences the biochemical stability of SOM (Plante et al., 2011; Kleber et al., 2005). Organisms in the rhizosphere decompose soil C to progressively smaller organic compounds which encourages reactivity with mineral surfaces and/or formation of aggregates, which promotes soil C recalcitrance (Schmidt et al., 2011). This and other factors ultimately lead to large differences in residence time of soil C across ecosystems (Lehmann and Kleber, 2015).

The molecular composition of SOM alone does not determine C residence time (Schmidt et al., 2011), but molecular structure can be a proxy for biochemical stability of soil C (Lehmann and Kleber, 2015). Upwards of 90% of all biomolecules in the SOM pool are proteins, lignin, carbohydrates, lipids, and aliphatic and carbonyl compounds (Nelson and Baldock, 2005), and the proportion of each differs across ecosystems, soil types, and climate regimes. Cellulose, a carbohydrate, is one of the least stable forms of soil C: it decomposes rapidly to smaller biopolymers which are then subject to a cohort of competing preservation and degradation factors (Broz et al., 2021). More resistant to decay are plant and microbial lipids (Silva et al., 2015) which, unlike cellulose, commonly persist in buried, ancient soils (paleosols) over geologic time scales (Marin-Spiotta et al., 2014), suggesting that NCS which promote increased abundances of soil lipids may effectively increase carbon permanence in soils. Similarly, pyrogenic C (char) exhibits biochemical stability over thousands of years in paleosols (Marin-Spiotta et al., 2014) and has been identified in Jurassic-age (Matthewmann et al., 2019) and Permian-age (Miller et al 1996) paleosols. Persistence of pyrogenic C in paleosols, potentially for hundreds of millions of years, suggests that pyrolysis of organic matter encourages carbon permanence over geological timescales. Natural climate solutions for agricultural areas include application of biochar (pyrogenic C) in crop systems, which can provide climate mitigation potential on par with the widespread restoration of coastal wetlands (Griscom et al., 2017), and potentially increase carbon permanence by 2-3 orders of magnitude relative to additions of more labile forms of soil organic matter as a part of NCS.

4. A next phase of understanding

In this work we introduced a novel but straightforward process for delineating areas of conservation or implementation of natural climate solutions. We use example ecosystems across the Pacific Northwest for a proof-of-concept of our proposed framework at local and regional scales. The recommended approaches for managing various landscapes for NCS can require a substantial commitment of finances and resources depending on current land management. Recent advances in ecological monitoring may serve to address this obstacle and allow for the next phase of understanding.

Widespread adoption of NCS that effectively increase carbon permanence should therefore be guided by prioritization at the local and regional scales. As a result, tailoring NCS implementation efforts to local and regional scales will maximize carbon sequestration and potentially serve to increase carbon permanence across landscapes. The approaches outlined in this work are interdisciplinary in nature, and thus future efforts to understand how to best implement natural climate solutions at the local and regional levels should involve stakeholders and researchers from across diverse disciplines and viewpoints.

Acknowledgements

We acknowledge the World Climate Research Programme's Working Group on Coupled Modelling, which is responsible for CMIP, and we thank the climate modeling groups for producing and making available their model output. For CMIP the U.S. Department of Energy's Program for Climate Model Diagnosis and Intercomparison provides coordinating support and led development of software infrastructure in partnership with the Global Organization for Earth System Science Portals."

Deanna!

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