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The spatiotemporal domains of natural climate solutions research and strategies for implementation in the Pacific Northwest, USA

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Natural climate solutions have been proposed as a way to mitigate climate change by removing CO_2 and other greenhouse gases from the atmosphere and increasing carbon storage in ecosystems. The adoption of such practices is required at large spatial and temporal scales, which means that local implementation across different land use and conservation sectors must be coordinated at landscape and regional levels. Here, we describe the spatiotemporal domains of research in the field of climate solutions and, as a first approximation, we use the Pacific Northwest (PNW) of the United States as a model system to evaluate the potential for coordinated implementations. By combining estimates of soil organic carbon stocks and CO_2 fluxes with projected changes in climate, we show how land use may be prioritized to improve carbon drawdown and permanence across multiple sectors at local to regional scales. Our consideration of geographical context acknowledges some of the ecological and social challenges of climate change mitigation efforts for the implementation of scalable solutions.

KEYWORDS

natural climate solutions, nature-based solutions, soil organic carbon, climate change, carbon persistence

Introduction

Natural Climate Solutions (NCS) are a set of strategies that involve the conservation, restoration, and sustainable management of ecosystems to enhance their ability to capture and store carbon dioxide from the atmosphere. These solutions leverage the natural processes of ecosystems to mitigate climate change by sequestering carbon and reducing greenhouse gas emissions. Natural climate solutions consist of land conservation, restoration, and management efforts that have potential to mitigate climate change by removing CO_2 and other greenhouse gasses from the atmosphere, as well as by increasing carbon (C) storage in terrestrial and nearshore ecosystems (e.g., Griscom et al., 2017). NCS can also achieve mitigation by avoiding emissions from changes in management or land use (e.g., manure acidification to avoid methane release, or protecting forests from conversion to avoid emissions from logging and subsequent decomposition of organic material) (Cook-Patton et al., 2021). Examples of NCS practices include reforestation and other forest management practices, regenerative agricultural practices such as cover cropping (Drever et al., 2021), biochar application, and ecosystem restoration (Graves et al., 2020). In the USA, many land-based NCS techniques are readily deployable and could mitigate about 21% of the annual atmospheric CO_2 emitted nationwide (Fargione et al., 2018).

However, the risks and benefits of NCS for people and ecosystems have yet to be fully assessed at local (tens of km^2) to regional (hundreds to thousands of km^2) scales. Simply put, these techniques will not work everywhere (Anderson et al., 2019), and in some cases, they may cause unanticipated release of CO_2 or a decrease in the mean residence time of carbon (Silva et al., 2022). Furthermore, many of the social and ecological factors that shape ecosystem dynamics and determine the overall success of potential NCS have yet to be quantified and incorporated into NCS frameworks (Silva, 2022). To advance research in this field, we explore the interactions between plants, soils and climate change. To enhance our understanding of the mechanisms that control carbon storage and release across ecosystems, we show how the majority of previously published studies of NCS often fail to address local and regional spatial scales as well as drawdown and storage over long (>100 years) temporal scales. Our objective is to build on existing knowledge to address data gaps and create an adaptable framework for prioritizing areas for NCS research and implementation.

Successfully implementing NCS requires a framework that is consistent at multiple spatial and temporal scales. Such a framework must consider local social and ecological factors that can be missed within large-scale efforts. For example, not all landscapes may be appropriate for NCS projects. Some have competing socioeconomic uses (e.g., forestry, agriculture, and/or indigenous land) (Hasegawa et al., 2018), some are projected to become too dry to sustain growing plant biomass (Marvin et al., 2023), and others are already rich in soil organic carbon (SOC) (Bossio et al., 2020). The variation in land use and resource availability constraints both the technical and realizable potential for C drawdown and permanence because no single NCS method is appropriate for all bioclimatic conditions (Baldocchi et al., 2018). Thus, scalable projects would require coordination of land use, conservation, and restoration efforts (Bossio et al., 2020; Silva, 2022). Here, we present a geographic-based framework for guiding NCS implementation using soil C stocks (a proxy for belowground sequestration) and projected climate change (a control on future drawdown potential) across multiple ecoregions in the Pacific Northwest (PNW) of the United States. In accordance with a "protect-manage-restore" approach to NCS implementation (Cook-Patton et al., 2021) that also considers future changes in climate (Marvin et al., 2023), the ecoregions considered here provide examples of how sector-specific NCS activities (e.g., forestry and agriculture) can be ranked relative to conservation and thus prioritized to inform decisions.

Within the exceptionally diverse bioclimatic, social, and ecological settings that exist in the Pacific Northwest, we chose ecosystems with exceptional NCS potential in the states of Oregon and Washington to serve as examples. We build on other examples of landscape prioritization for C sequestration (Cook-Patton et al., 2021; Law et al., 2022) and socioecological benefits through carbon markets (Aoyama et al., 2022; Bomfim et al., 2022), conservation and management that influence how carbon drawdown efforts affect the water cycle (Maxwell et al., 2018) as well as plant-soil feedbacks that influence nutrient use and carbon stabilization (Silva and Lambers, 2020) and lessons from other model regions where "fundamental mechanisms can be identified, understood, and tested" (Vitousek and Loope, 1987). The resulting NCS priority areas identified below serve as a delineation of high-priority NCS locations, as a first approximation toward a framework for implementation at regional to local scales.

Methods

Inspired by previous studies of spatial and temporal domains in ecology (Estes et al., 2018), we set out to conduct a systematic meta-analysis using data from a literature review that covers most NCS studies published through 2023. To assess the general representation of spatial and temporal scales that have been considered in the NCS literature, we gathered data from previously published studies about NCS (Figure 1). We reviewed 55 studies published from 2011 to 2023 that contained the keywords "Natural Climate Solutions" or "Nature-based Solutions". These studies used either a field-based and/or a modeling approach for estimating CO_2 sequestration potential of NCS, which allowed for approximating the spatial and temporal extent considered in each study (Supplementary material). 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 modeled 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 ($GtCO_2$ eq. yr^{-1}) of a NCS, we approximated the temporal extent of the study to be 20–100 years. As many studies reported C permanence on temporal scales of ∼20, 50, or 100 years (e.g., predictions for the years 2040, 2070, or 2100), we considered a bin of 20–100 years to accommodate this range. It should be noted that this approach considers a large temporal scale (e.g., 80 years) that has large variations in mitigation potential and associated uncertainties (e.g., Griscom et al., 2017).

A large proportion of the papers reviewed propose various NCS at a national or global scale, and thus precise estimates on the scales of NCS areas have large associated uncertainties. For example, many NCS papers cover a range of practices that vary widely in their spatial extent. We dealt with this variability by considering the spatial scales of all of the proposed NCS in each paper, not for each

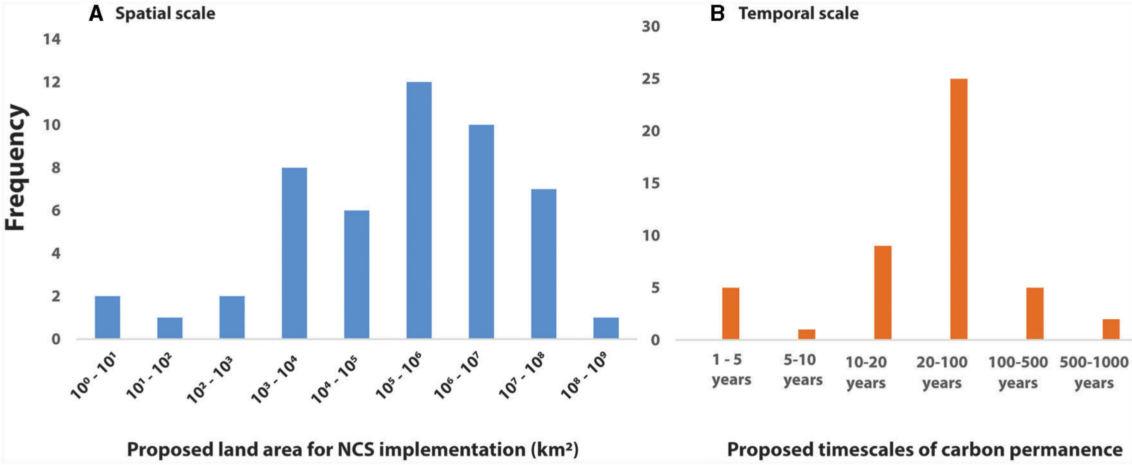

FIGURE 1
Data from previously published work showing the spatial and temporal domains of natural climate solutions (NCS) research. **(A, B)** Histograms of **(A)** the spatial scale of NCS implementation and **(B)** temporal scales of carbon permanence collectively gathered from a survey of 30 studies focused on natural climate solutions. Local scales are defined here as ~10^1-10^2 km^2 and regional scales are ~10^2-10^3 km^2.

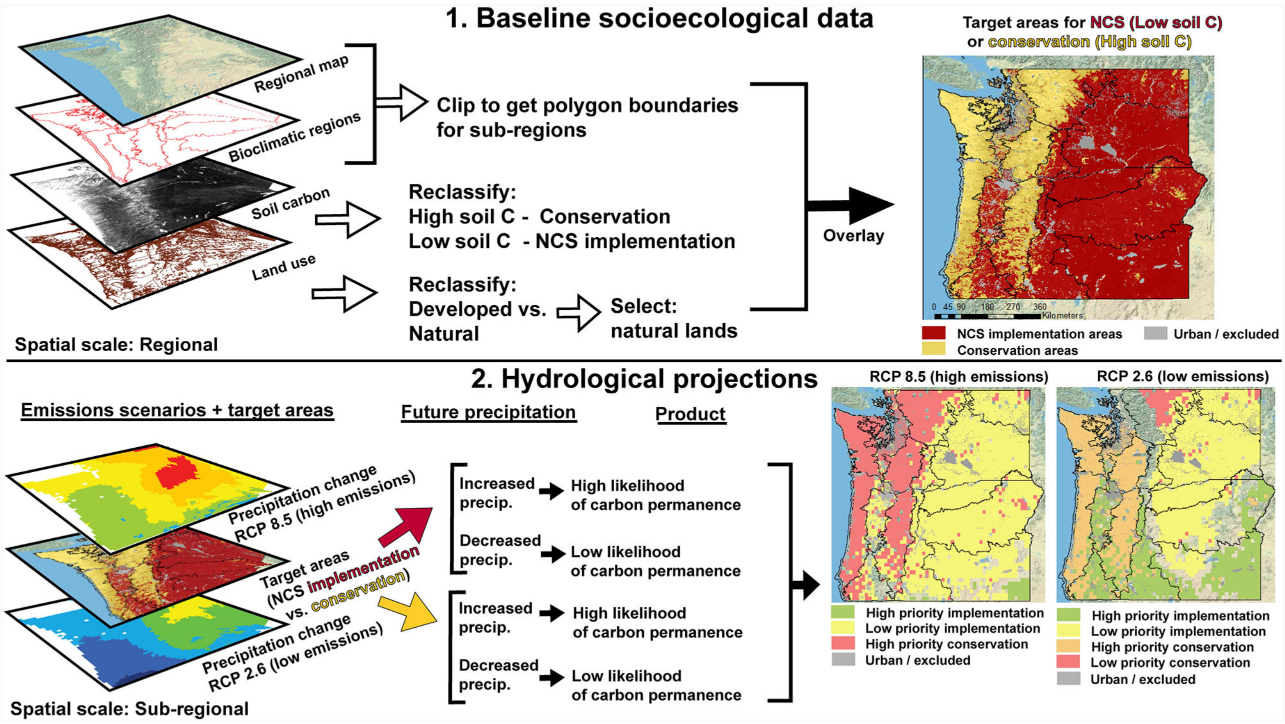

FIGURE 2
A proposed framework for prioritization of natural climate solutions (NCS) implementation areas or conservation across regional and local spatial scales. Step 1 uses baseline data of a socio-ecological context including bioclimatic zones, soil carbon stocks, and land use to identify target areas 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.

individual/specific practice (e.g., the modeling of GHG drawdown from NCS was at a national or global level).

We next used available geospatial data that caters to local and/or regional scales to prioritize areas for NCS implementation or conservation, ensuring a replicable approach (Figure 2). Here, we consider conservation as distinct from NCS, which include management and restoration. We acknowledge that most literature on NCS considers "conservation" as a type of NCS, but the ecoregions considered here provide examples of how sector-specific NCS activities (e.g., forestry and agriculture) can be ranked relative to conservation and thus prioritized to inform decisions.

Bioclimatic regions are defined here by Washington and Oregon state Level III ecoregions, defined by 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 (U. S. EPA, 2020). Terrestrial baseline soil carbon was based on 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.

Finally, we propose a combination of biophysical and land use maps that consider existing carbon baseline data into opportunity maps for drawdown and permanence under future climates. To this end, we used the downscaled version of CMIP5 multi-model ensemble output, which utilizes 1/8° resolution bias corrected precipitation projections (Reclamation, 2013) that were produced by running period analyses using the MIROC5 (Model for Interdisciplinary Research on Climate), applying 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. The RCP2.6 scenario assumes substantial changes in energy use, but a large increase in global cropland area. In contrast, the RCP8.5 scenario represents a 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.

Results and discussion

Observational scales of natural climate solutions

A compilation of 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 modeling 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 focus on such large spatial scales and

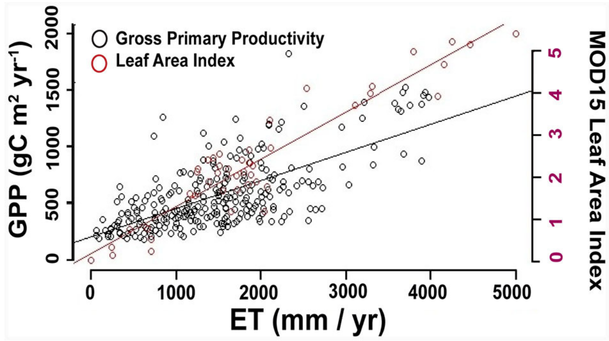

FIGURE 3
Relationships between evapotranspiration (ET), gross primary productivity (GPP), and leaf area index (LAI) (the "carbon-water tradeoff"). Data for the GPP and ET relationship were adapted from Baldocchi and Penuelas (2019). The GPP and ET values represent annual averages of eddy covariance measurements from the FLUXNET 2015 dataset (Pastorello et al., 2020) that includes 155 global flux towers (Supplementary Table 1) (Chu et al., 2021). 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 to 2008 from MODIS product cutouts surrounding 21 FLUXNET towers (Supplementary Table S2), and include tree LAI for agricultural settings, represented by the equation $LAI = 0.76 + 0.00153\,(ET)$.

may fail to consider local and regional challenges to NCS (Mu et al., 2019). Similarly, the focus of carbon permanence on short temporal scales (e.g., 20 years) may not adequately consider the long-term C sequestration potential of agricultural NCS such as biochar application or enhanced mineral weathering, which may encourage C permanence over geologic time scales (e.g., 500–1,000 years), further discussed in the "Carbon Permanence" Section (Section Discussion).

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 modeled 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. Our estimation errors are therefore incomplete and should be expanded in future studies. Despite this limitation, our analysis provides a general frame of reference to interpret the most common observations in the consideration of NCS implementation and monitoring. Results in Figure 1 highlight the need for a focus on local to regional scales that consider carbon permanence on time scales longer than 100 years. In the following sections, we address the opportunities and tradeoffs for NCS implementation across these spatial and temporal scales.

Guiding tradeoffs for NCS prioritization

This geographical framework for NCS implementation is focused on addressing two challenges within the process of research and implementation: (1) delineating locations that present opportunities for carbon drawdown, and (2) identifying areas that would benefit more from conservation than management for NCS implementation (Figure 2). Our suggestion for developing NCS strategies is guided by the tradeoffs that are inherent in ecosystem mining of CO_2 from the atmosphere (Baldocchi et al., 2018). We use the carbon-water relationship (Figure 3) 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.

The assimilation of carbon through vegetation is physically and biologically tied to water use and plant morphology (e.g., Silva and Lambers, 2020). Gross primary production (GPP) scales with evapotranspiration (ET) and Leaf Area Index (LAI) (Figure 3) quasi-linearly, showing 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 loss through ET (Liles et al., 2019; Maxwell and Silva, 2020; Segura et al., 2021; Law et al., 2022). The scaling of GPP with ET, which varies in tandem with LAI within a given range of vapor pressure deficit, suggests that ecosystem functions relevant to NCS are strongly tied to ecosystem structure composition and climate (Hudiburg et al., 2013).

The significant variability about the mean regression line (Figure 3) is an important indicator of differences in ecosystem scale water use efficiency toward carbon drawdown. A reasonable interpretation of this variability is that sites or ecosystems represented by points falling below the regression line are more efficient with respect to water use than points falling above the regression line. Species composition and ecosystem structure are primary sources of variation in the relationship between GPP and ET (e.g., Migliavacca et al., 2021), and the regression between LAI and GPP in part explains the role and water cost of assimilating CO_2 in diverse communities and ecosystems. However, this relationship does not fully explain the variability in the ET—GPP data. We suggest that future studies should prioritize characterizing the sources of this variability to better support efforts to optimize the balance between water loss and carbon gain through NCS implementation. Further, we emphasize that successful implementation of NCS management techniques will require optimizing CO_2 fixation per unit of water lost to evapotranspiration, which relies upon characterizing the sources of variation in the carbon-water tradeoff for different ecosystems to inform conservation, management, or restoration decisions.

The role of plant species composition, morphology, and ecosystem structure in defining carbon-water tradeoffs can be extended to multiple spatiotemporal scales. For example, the evolution of vein density in leaves and wood transport structure can be used to reconstruct past and predict future canopy conductance and ecosystem responses to vapor pressure deficit and climate variability (e.g., Boyce et al., 2009; Earles et al., 2016; Sperling et al., 2017).

Within the Pacific Northwest, the primary forest types that are considered in NCS plans occur in mesic and dry landscapes. 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 4). Ecophysiological differences cause inter- and intraspecific variation in growth rates throughout these forests (e.g., Hudiburg et al., 2013). In general, trees growing in mesic conditions have higher GPP and lower WUE (water-use efficiency) than their more arid counterparts (Ruzicka et al., 2017). Driven by biotic and abiotic conditions, water use efficiency varies across species, space, and time and 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.

Agricultural areas in the PNW and elsewhere present additional uncertainties for estimating the carbon-water tradeoff. Agricultural areas typically exhibit low Leaf Area Index (LAI) for trees or shrubs, except in cases like orchards, silvopastoral systems, or agroforestry (e.g., He et al., 2020). However, crops in these areas often have high LAI for non-woody species (Du et al., 2022). A key overall distinction is the significant seasonal fluctuation of agricultural LAI, which perhaps limits its role as a stable and substantial carbon sink.

This variation in water-carbon relations is of significant consequence to the applicability of NCS implementation at a regional scale (Figure 4). In Oregon, the GPP of a mesic Douglas fir-dominated wet forest is 2,590 $gCm^{-2}yr^{-1}$ vs. 1,631 $gCm^{-2}yr^{-1}$ in dry forests of ponderosa pine approximately 160 km to the east (Kwon et al., 2018). Critically, the intrinsic WUE of semi-arid ponderosa pine forests is estimated to be approximately four times greater than in mesic Douglas-fir forests (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 vs. higher intrinsic WUE in drier sites (Ruzicka et al., 2017). While moisture availability is primary control on plant growth, additional parameters such as geomorphological setting, soil nutrient availability, growing season length further constrain NCS potential (Maxwell et al., 2018). Our preliminary maps (Figure 5) point to areas where NCS implementation is desirable, such as across the Willamette Valley in Oregon, but there are additional considerations that we believe will ultimately determine the effectiveness of NCS. These include considerations of past and current soil conditions and land cover type, which has implications for baseline conditions and for increasing carbon permanence across landscapes, especially across PNW agricultural areas (e.g., Waldo et al., 2016) where precipitation is modeled to increase under both RCP pathways (Karimi et al., 2017; Mu et al., 2019). Prioritization of conservation or management for NCS across several ecosystems of the PNW "natural laboratory" (Figures 4, 5) are discussed in the following paragraphs.

Wet forest

Globally, carbon- and water-rich habitats are often co-located and spatiotemporally correlated (REF). Consistent with this global pattern, some of Earth's most carbon-rich forests lay within the wet temperate zone in the humid PNW (Figures 4, 5). The measured carbon densities from PNW 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

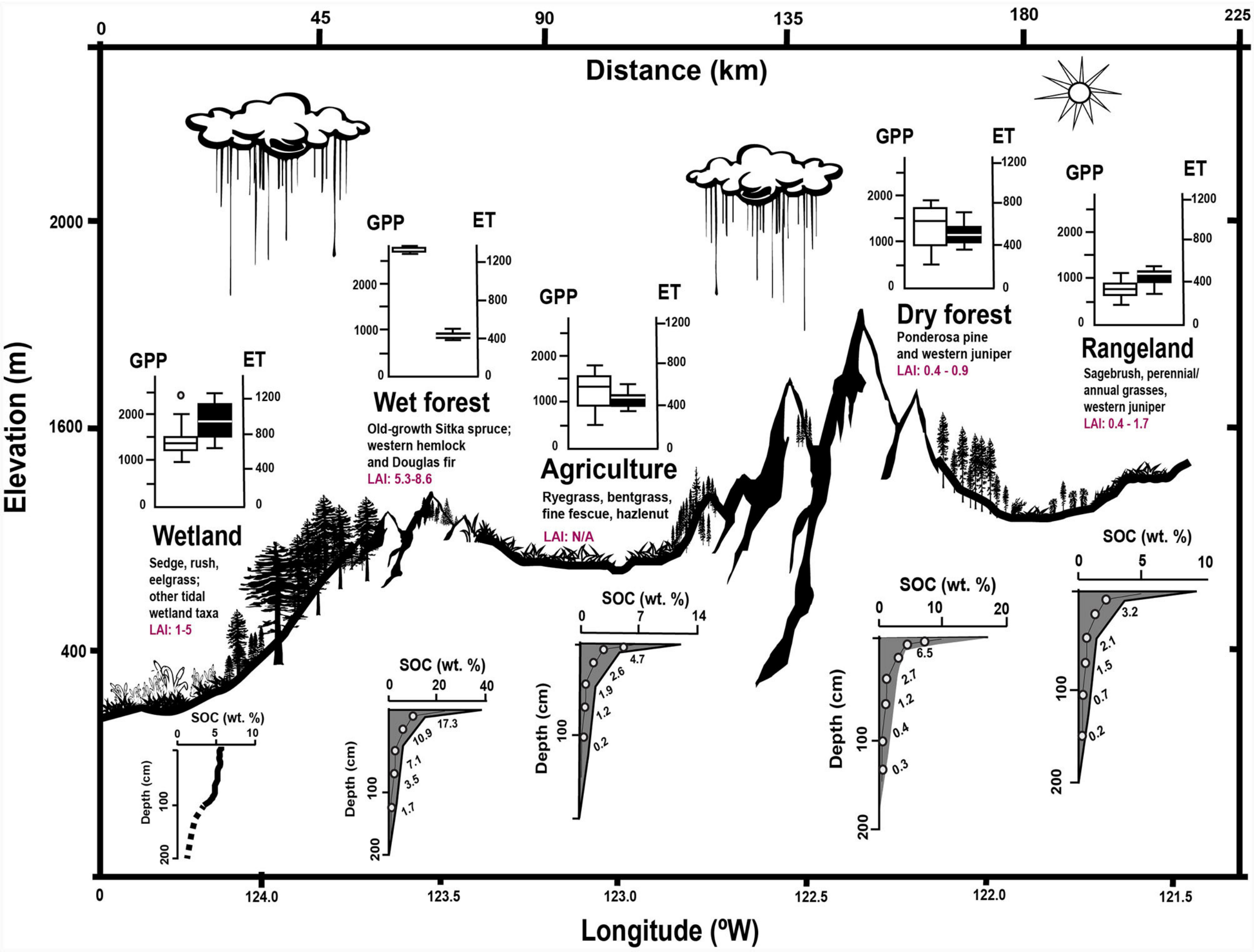

FIGURE 4
Carbon-water tradeoffs across a Pacific Northwest transect. Boxplots show the carbon-water tradeoff (Gross Primary Productivity [GPP] and evapotranspiration [ET]) across ecosystems (data from Pacific Northwest eddy covariance sites in Baldocchi and Penuelas, 2019 and Kwon et al., 2018 [wet forest data]). Leaf area index (LAI) data were from Bonan (2015); agricultural LAI was defined as N/A because of large seasonal LAI variations typical of agricultural settings (e.g., Du et al., 2022). Modeled 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. Modeled 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. 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 modeled from 100 to 200 cm (dotted line).

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 because 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.

In Figure 4 we show that coastal forests, as one example of wet forests, have a similar range of GPP and ET values, where the cost of maintaining a high level of GPP requires more ET than other ecosystems. This pattern is not ubiquitous for all wet forests, however, and it varies upon management history and microclimate. The coastal forest ecoregion 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 ecoregions 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 and 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 water-carbon tradeoffs. The spatial extent of NCS conservation areas across wet forests of western Oregon presented here (Figure 5) overlap with previously published regional frameworks for maximizing C storage and preserving biodiversity through the establishment of strategic forest reserves in Oregon (Law et al., 2022).

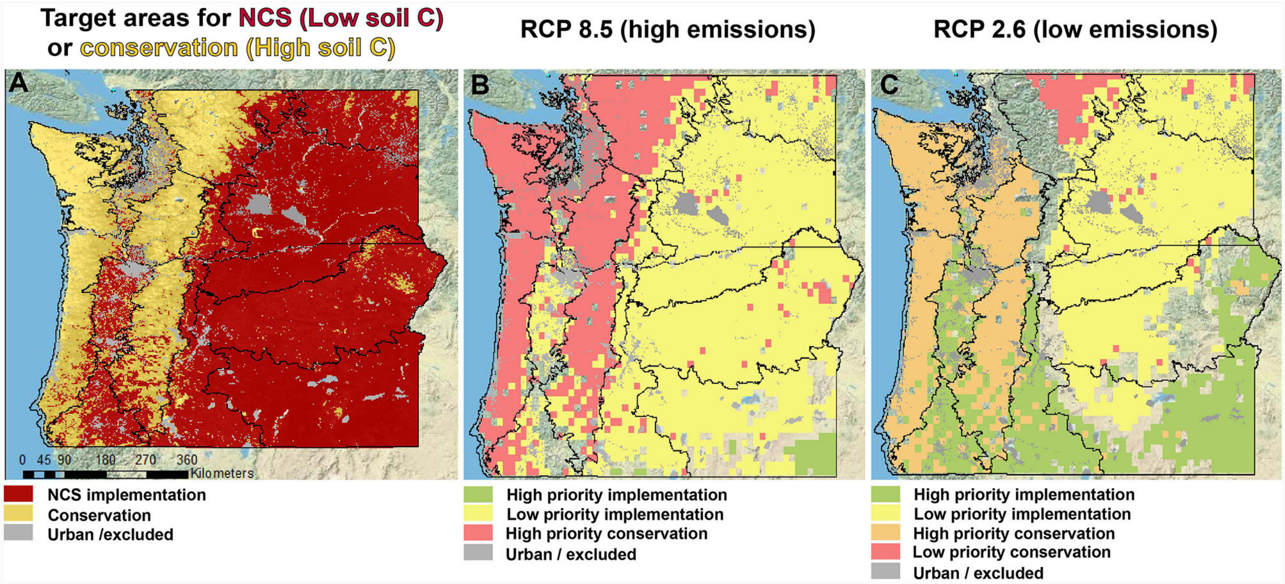

FIGURE 5
Data products for NCS prioritization areas in the Pacific Northwest considering a carbon-water tradeoff under different emissions scenarios. **(A)** Consideration of bioclimatic zones, soil carbon stocks, and land use to identify target areas for NCS implementation, where soil C is low (<5 wt. % soil organic carbon [SOC]) or high (<5 wt. % SOC); **(B, C)** combine target areas (red) from **(A)** and incorporate future projections of precipitation using both high and low emission scenarios to rank areas by priority, where "high priority implementation" areas have low SOC and projected increases in precipitation.

With the goal of increasing carbon drawdown and permanence, we suggest that NCS in coastal PNW forests of Oregon and Washington (Figure 5) should be focused on conserving existing carbon in the form soil organic carbon and, whenever possible, expanding ecological restoration and forestry for sustainable timber management and extended rotations. Ecological forestry favors 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). Our suggestion is consistent with the guiding principles of Ecological Forestry, which has been favored by the Northwest Forest Plan (NWFP) as a way to reduce harvesting of old-growth forests, while also allowing for clearcutting and thinning in certain contexts. Furthermore, NCS recommendations for conservation of wet forests with high baseline aboveground and soil C (Figures 4, 5) are consistent with previous work in the PNW calling for protection of large trees (>51 cm diameter), which dominate aboveground carbon storage in both wet and dry forests (Mildrexler et al., 2020). Such trees can account for just 3% of all stems in PNW forests but hold 42% of total aboveground carbon (Mildrexler et al., 2023).

Our analysis also suggests that large scale shifts from mature/old forests toward dense tree plantations might be unsustainable under future climates because it could compromise the provision of water for people and ecosystems, increasing the risk of catastrophic wildfire and related carbon emissions, as already observed in some regions within the PNW and elsewhere (Perry and Jones, 2017; Jerrett et al., 2022). To address this issue, we suggest a greater emphasis on integrated landscape management practices intended to preserve and increase baseline carbon stocks. These may include longer forestry rotations with riparian buffers (Segura et al., 2021), which have some of the highest carbon sequestration potential in the region (Graves et al., 2020), with the possibility of including community-led prescribed fires to decrease the risk of wildfire (Lake et al., 2017).

Dry forest

The dry forests of the eastern Cascades, Klamath, and interior mountains are dominated by ponderosa pine (Figure 4) 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. 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 change 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 going forward (Liang et al., 2018).

East of the Cascades, landscapes tend to support less productive, drier conifer forests (Figure 4), which are typically subject to frequent fires of low to mixed severity. Due to the variation in forest productivity, land use, and historical fire patterns, differences in Non-Industrial Private Forest (NIPF) management and owner responses are expected across the ecoregions. This expectation led to the necessity of using a stratified approach to prioritize landscape sampling and design of NCS projects.

Fuel is the most significant control on fire severity in these forests, followed by fire weather, climate, and topography, respectively (Parks et al., 2018). As a consequence, land management decisions have critical implications for the 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).

In this way, fuels and fire management planning is the Natural Climate Solution. All NCS activities must consider the preexisting fire regime to ensure the long-term effectiveness of C drawdown, favoring riparian restoration and conservation efforts over expanding tree plantations within landscapes that may be primed to burn at high 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) and is applicable to our study area in the PNW (Figure 4). 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 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). Our analysis suggests that current bioclimatic conditions and modeled changes in precipitation (Figure 5) could serve as a blueprint to understanding and predicting the success of NCS projects under predominant fire regime and help to inform appropriate landscape specific conservation and/or management actions (Figure 2). This approach is aligned with delineations at local to regional scales of dry forest areas in Oregon and Washington where fire could be reintroduced to thin understory vegetation (Law et al., 2023).

Tidal wetlands

Tidal wetlands are valuable carbon sinks present within the Coast Range ecoregion (Figure 4). 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 ∼5,200 ha of degraded area (Graves et al., 2020). Despite the limited geographic extent, tidal 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) (Figure 4). 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 during flood events can result in lower CO_2 emissions and unchanged CH_4 emissions (Negandhi et al., 2019).

The carbon-water tradeoff for NCS implementation in tidal wetlands is distinct from other ecosystems considered in this work (Figure 4). For example, high ET may not be as detrimental to this aquatic landscape as it is to other ecosystems in our model region (Figure 4). Instead, anthropogenic activities 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).

Varying soil organic carbon stock in PNW estuaries also provides opportunities and challenges for prioritizing NCS implementation (Figure 5). In general, carbon storage in estuaries varies by latitude and estuary size (Hinson et al., 2019). However, conservation and management decisions can greatly improve NCS potential in these systems. For example, a spatial analysis combining extensive sampling across salinity gradients and management regimes (disturbed, restored, and reference) in 22 wetland sites across two Oregon estuaries, demonstrates that it is possible to estimate GHG fluxes in both existing and former wetlands in the region, from a small set of environmental data combined with past and current land use data (Schultz et al., 2023). This requires collecting environmental data that accurately captures the variability of the site and spans a sufficient time frame to account for seasonal changes (e.g., Pham et al., 2019), ideally covering at least a full year and employing a widespread carbon flux sampling method that utilizes machine-learning. Adopting this method could offer a cost-effective way to assess the viability of wetland restoration as a strategy for climate mitigation. 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 in the PNW should be preserved, whereas areas that are bare but capable of hosting native species, such as eelgrass (*Zostera marina)*, should be targeted for conservation and restoration.

Similarly, seagrass conservation and restoration efforts can be used to increase carbon capture with multiple benefits. Prioritization based on pre- and post-event disturbance data collection has been proposed to leverage long-term monitoring with new machine learning approaches for developing funding streams to support flexible decisions and prepare for extreme future conditions (Aoki et al., 2022). Although seagrass restoration is generally thought to have low success rates in the PNW (Fonseca et al., 1988; Stamey, 2004), compelling results have been obtained in areas where removing dikes to reinstate tidal wetlands can be implemented to encourage seagrass expansion and eradication of invasive species (Herrick and Wolf, 2005). Indeed restoration and conservation of tidal wetlands has many benefits when compared with business-as-usual use of such ecosystems, including implications for net carbon storage following disturbance events, but many questions remain with respect to carbon source and permanence in tidal wetlands. How much of the carbon is sequestered *in situ* vs. deposited from other systems? How long will the sequestered carbon be stored? We suggest that these and other questions should be considered in future efforts to prioritize NCS implementation especially across areas containing significant terrestrial and aquatic carbon sinks.

Carbon permanence

The long-term success of NCS ultimately depends on carbon permanence, defined as the mean residence time of sequestered carbon across landscapes (Figure 1). 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. (2020) to limit warming to 1.5°C above preindustrial levels. Natural climate solutions have the potential to sequester between 100 and 550 Gt CO_2 between 2018 and 2100 at a cost of <\$100 per ton carbon (Griscom et al., 2017), but ensuring that sequestered C is sufficiently "permanent" will determine the ultimate success of NCS. For example, previously proposed approaches to increase carbon stocks on land through afforestation, even when considering soil type (Bastin et al., 2019) generally do not emphasize baseline soil organic carbon stocks, which in some cases can be higher in unmanaged savannas and grasslands than in planted forests. Thus, if implemented through uniform tree plantings across landscapes, NCS efforts could result in unintentional liberation of C from areas with high baseline soil C (Figure 5A). Inherent differences in soil C stocks across ecosystems demand landscape-specific solutions for encouraging long-term soil C sequestration. For example, geomorphological and ecological context can be used for better predicting baseline conditions as well as projecting the potential for additional landscape carbon capture (Roering et al., 2023).

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 (Figure 5) 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), including those of the PNW study region (Figure 4). Sorption of organic molecules to the surfaces of minerals and amorphous colloids influences the biochemical stability of SOM (Kleber et al., 2005; Plante et al., 2011). 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 permanence (Schmidt et al., 2011). This and other factors ultimately lead to large differences in residence time of soil C across ecosystems in the PNW and elsewhere (Lehmann and Kleber, 2015) and many mysteries remain to be solved with respect to the stability of different SOM fractions, their potential sources, and permanence.

The molecular composition of SOM alone does not determine carbon permanence (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. Plant and microbial lipids are more recalcitrant relative to cellulose (Silva et al., 2015) and can persist in buried, ancient soils (paleosols) for tens of thousands of years or longer (Marin-Spiotta et al., 2014), suggesting that NCS management strategies that 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 early Holocene paleosols (Marin-Spiotta et al., 2014) and is found in much older paleosols (Retallack, 2019) from the late Jurassic (∼180 million years old) (Matthewmann et al., 2019) and late Permian (∼290 million years old) (Miller et al., 1996).

The persistence of pyrogenic C in paleosols suggests that pyrolysis of organic matter could encourage carbon permanence over long (e.g., geological) timescales (Broz, 2020). For this reason, NCS activities for agricultural areas in the PNW (Waldo et al., 2016) can include application of biochar (pyrogenic C) to cropping systems, which can provide climate mitigation potential on par with the widespread restoration of coastal wetlands (Griscom et al., 2017). Such techniques could potentially increase carbon permanence by two to three orders of magnitude relative to additions of more labile forms of soil organic matter (e.g., cellulose). Addition of biochar to natural and agricultural soils such as those in the Willamette Valley in Oregon (Figure 5) may therefore be an effective technique for increasing carbon permanence in the PNW region and elsewhere.

A next phase of understanding

In this study we explore the potential of existing baseline data combined with climate projections to orient NCS through coordinated land conservation or management. We use example ecosystems across the PNW to illustrate how local and regional decisions might reduce risks and improve benefits of NCS activities. We caution that concrete implementation plans must go beyond the general examples proposed here to consider social impacts and to evaluate ecologically appropriate contexts under current and projected bioclimatic conditions, ideally using a co-production of knowledge approach to quantify risks and benefits of widespread adoption in vulnerable communities. The recommended approaches for managing or conserving landscapes for NCS would require a substantial commitment of finances and resources, which goes beyond the scope of ecosystem processes discussed here. Recent advances in ecological monitoring may serve to address this obstacle and allow for the next phase of understanding in which bottom-up community-led NCS adoption can effectively increase carbon drawdown and permanence at local and regional scales.

Adoption of this framework for prioritizing NCS implementation yields three primary benefits: (1) By balancing NCS potential with carbon permanence potential, estimates of carbon sequestration take into consideration both the short-term and long-term dynamics of NCS implementation within the context of climatic change. (2) Adopting a scalable approach to parameterizing NCS potential reduces uncertainty in national sector-based estimates of carbon sequestration by considering variability in sequestration rates introduced by local geographic features, climatic conditions, and plant assemblages. (3) The flexibility of this framework aids prioritization of NCS implementation at a regional scale, while also enabling land managers to consider the suitability and potential tradeoffs of NCS implementation at the local and landscape-scales.

Future efforts should also include consideration of inorganic carbon sinks, such as enhanced weathering projects, which could have greater carbon sequestration potential than the biological processes described above globally. For example, agriculture could sequester 4.1 Pg CO_2 eq. year^{-1}; ecological restoration (e.g., protection and sustainable management of forests, savannas, and grasslands) could sequester 7.3 Pg CO_2 eq. year^{-1}, whereas enhanced weathering could sequester 1–100 Pg CO_2 eq. year^{-1} although with much larger uncertainties (Shukla et al., 2019). Tailoring global NCS estimates and implementation efforts to specific regions with diverse social and ecological contexts is a challenge that requires further interdisciplinary research. Thus, future implementation efforts should involve stakeholders and researchers from diverse disciplines, sectors, and socioeconomic backgrounds.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

OC: Conceptualization, Data curation, Formal analysis, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. AB: Conceptualization, Methodology, Visualization, Writing – original draft, Writing – review & editing. EL: Conceptualization, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. MF: Conceptualization, Methodology, Software, Visualization, Writing – original draft, Writing – review & editing. RA: Conceptualization, Visualization, Writing – original draft, Writing – review & editing. LS: Formal analysis, Methodology, Supervision, Validation, Visualization, Writing – original draft.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fclim.2024.1273632/full#supplementary-material

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