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Pedoecological Modeling to Guide Forest Restoration using Ecological Site Descriptions

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The U.S. Department of Agriculture (USDA)-Natural Resources Conservation Service (NRCS) uses an ecological site description (ESD) framework to help incorporate interactions between local soil, climate, flora, fauna, and humans into schema for land management decision-making. We demonstrate ESD and digital soil mapping tools to (i) estimate potential O horizon carbon (C) stock accumulation from restoring alternative ecological states in high-elevation forests of the central Appalachian Mountains in West Virginia (WV), USA, and (ii) map areas in alternative ecological states that can be targeted for restoration. This region was extensively disturbed by clear-cut harvests and related fires during the 1880s through 1930s. We combined spodic soil property maps, recently linked to historic red spruce–eastern hemlock (*Picea rubens–Tsuga canadensis*) forest communities, with current forest inventories to provide guidance for restoration to a historic reference state. This allowed mapping of alternative hardwood states within areas of the spodic shale uplands conifer forest (SCF) ecological site, which is mapped along the regional conifer-hardwood transition of the central Appalachian Mountains. Plots examined in these areas suggest that many of the spruce-hemlock dominated stands in WV converted to a hardwood state by historic disturbance have lost at least 10 cm of O horizon thickness, and possibly much more. Based on this 10 cm estimate, we calculate that at least 3.74 to 6.62 Tg of C were lost from areas above 880 m elevation in WV due to historic disturbance of O horizons, and that much of these stocks and related ecosystem functions could potentially be restored within 100 yr under focused management, but more practical scenarios would likely require closer to 200 yr.

Abbreviations: CNIMP, conifer importance; dbh, diameter at breast height; ESD, ecological site description; IMP, Importance; MNF, Monongahela National Forest; SCF, spodic shale uplands conifer forest; WV, West Virginia.

Soils are a dynamic interface between abiotic and biotic drivers and the Earth's crust. In soil science this has been conceptualized as a state factor model where the state or properties of a soil are a result of interactions between climate, organisms, relief, and parent material over time (Dokuchaev, 1999; Jenny, 1941). The state factor model evolved to an ecosystem level model where soils and organisms have some parallel drivers, but also interact strongly (Eq. [1], Amundson and Jenny, 1997; Jenny, 1961; Jenny, 1980).

$$l, s, v, a = f(L_0, P_x, t) \quad [1]$$

The dependent factors in this case include ecosystem properties (l), soil properties (s), vegetation (v), and animals (a). The related state factors in an ecosystem based approach include the initial state (L_0) and external potentials (P_x), and time (t). Initial state L_0 includes the parent material (bedrock or substrate), initial relief, and water table. Climate and organisms are grouped as the P_x variable, which represent the primary energy sources that drive processes (Jenny, 1961). Amundson and Jenny (1991,

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1997) have introduced these conceptual models into ecological sciences, with humans included in the factorial equation. Soils bear the imprint and help record the story of organisms—especially humans—and the climate as pedomemory signals in biogeochemical and physical properties that can be valuable in understanding the history of sites (Lin, 2011; Monger et al., 2015; Monger and Rachal, 2013; Nauman et al., 2015; Phillips and Marion, 2004; Targulian and Goryachkin, 2004).

Within the context of the multi-factorial soil system, understanding the relationship between soils and associated ecosystems has been incorporated into different land management schemes including Landfire (Rollins, 2009), Terrestrial Ecological Units (Winthers et al., 2005), and Ecological Sites (USDA-NRCS, 2014; Caudle et al., 2013), among others. We focus on ESDs, because of their use in connection to soil components mapped nationwide in the USA by the NRCS. Ecological site descriptions have been extensively used to help land managers employ sound science in taking appropriate management actions in the rangelands of the USA (Bestelmeyer et al., 2011, 2009; Briske et al., 2005; Caudle et al., 2013; Grazing Lands Technology Institute, 2003; Herrick et al., 2006). Recently the NRCS, the government agency behind most ESD development, has put more emphasis into applying this framework in the eastern USA and has released a new handbook to help incorporate appropriate methods for developing ESDs for eastern forested systems (USDA-NRCS, 2014). The conceptual importance of ESDs is in recognition of connections and interactions between groups of floral and faunal species and specific soil properties. Put more specifically by the USDA-NRCS (2014), an ecological site is "a distinctive kind of land based on recurring soil, landform, geological, and climate characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its ability to respond similarly to management actions and natural disturbances." Ecological sites are a framework of organizing areas with similar ecosystem factors (Eq. [1]) that can drive soil-biota processes down pathways resulting in specific soil morphologic expression, and can provide an insightful narrative into the history of the land.

Podzolization Pathway and Soil Organic Carbon

In WV (USA) there has been debate over whether high elevation Spodosols and associated spruce-hemlock forest communities extend into lower elevation shale geologies due to contrasting reporting in soil mapping projects (Losche and Beverage, 1967; USDA-SCS and USDA-FS, 1982; Williams and Fridley, 1931). Spodosols are a result of soil development along the podzolization pathway (Lundström et al., 2000a, 2000b; Sauer et al., 2007), which often relates to forest species composition (Miles, 1985; Willis et al., 1997). There also is often formation of thick surface O horizons at the soil surface, especially in moist conifer systems (Hix and Barnes, 1984; Lietzke and McGuire, 1987; Lundström et al., 2000a).

Much of the organic C in Spodosols can be lost in 30 to 100 yr just by converting cool, moist acidic conifer forest stands to differing species compositions (prairie or hardwood) that favor more decomposition (Barrett and Schaetzl, 1998; Hix and Barnes, 1984; Hole, 1975; Miles, 1985). This is most prominent in the forest floor O horizons, which get thinner in the conversion (Barrett and Schaetzl, 1998; Hix and Barnes, 1984; Miles, 1985). Studies have also shown that conversion from hardwood forests (e.g., *Quercus spp., Betula spp.*, and *Fagus spp.*) to Norway spruce (*Picea abies*) and/or Scots pine (*Pinus sylvestris*) causes O horizon buildup and increased podzolization (Herbauts and Buyl, 1981; Miles, 1985; Ranger and Nys, 1994; Sohet et al., 1988). Forest common garden plot studies that isolate tree species on individual plots have also shown a contrast between species that promote base cation activity and heterotrophic organic matter decomposition (e.g., *Acer spp.* and *Tilia spp.*), and those that favor acidic Al and Fe activity (e.g., *Pinus spp. and Larix decidua*) which were associated with less decomposition of soil organic matter (Hobbie et al., 2007). In this same garden plot, higher tree litter Ca content appeared to control pH, decomposition, and stimulate earthworm activity, which resulted in less forest floor mass (Hobbie et al., 2006; Reich et al., 2005). Hobbie et al. (2006) also showed that spruce and fir species were associated with lower mean annual soil temperatures and less litter decomposition. Although influential general differences in litter chemistry were seen between angiosperms (basic) and gymnosperms (acidic), these studies showed significant variation within these groups of species. Another recent common garden study in New York showed a similar influence of worms under northern red oak (*Quercus rubra.*) and sugar maple (*Acer saccharum*), but not under Norway spruce which formed deeper forest floors (Melvin and Goodale, 2013). Although Ca^{2+} was similar under all three species, pH was lower under the spruce, suggesting that base cation activity might not be the only factor to examine. Overall, these studies tell a story where heterotrophic forest litter decomposition and O horizon accumulation are intricately linked with each tree species present at a site.

Forest History and Ecological Change in Red Spruce Systems

Human disturbance, pollution, and climate change are thought to have contracted regional red spruce populations, but are somewhat hard causal factors to distinguish (Hamburg and Cogbill, 1988). A warming climate is pushing cooler subalpine conifer ecosystem species like red spruce higher in elevation and higher in latitude, putting large pools of soil organic C at risk for further atmospheric release (Lal, 2005). Acid deposition damage to red spruce health has also been studied (Adams and Eagar, 1992; Hornbeck and Smith, 1985; Johnson, 1983), but might be hard to separate from the impact of a changing climate, overall warming (Hamburg and Cogbill, 1988) and extensive historic clear cutting and associated fires and pest outbreaks (Clarkson, 1964; Hopkins, 1899; Korstian, 1937; Minckler, 1945; Pauley, 2008; Pielke, 1981; Stephenson and Clovis, 1983). Indeed, red spruce is projected by different climate change scenarios to disappear from WV by the end of the century (Butler et al., 2015; Byers et al., 2010). However, there are signs that red spruce is recovering from historic disturbance and could be further restored despite climate change (Nowacki et al., 2010; Rentch et al., 2007; Rentch et al., 2010; Rollins et al., 2010). At this time, its future remains uncertain.

In WV, historical accounts indicate that the current extent (~20,000 ha) of red spruce forest communities is significantly reduced from its range before extensive logging and fires between 1800 and 1930 (~200,000 ha; Clarkson, 1964, 1993; Hopkins, 1899; Minckler, 1945; Nowacki and Wendt, 2010; Pauley, 2008; Pielke, 1981). Maximum entropy and logistic regression modeling efforts have similarly shown that the suitable habitat for red spruce in WV is much more extensive than current distributions reflect (Beane et al., 2013; Byers et al., 2010; Nowacki and Wendt, 2010). These studies, along with broader analysis of red spruce habitat (Nowacki et al., 2010) show temperature and precipitation as the main controls on extent. However, recent work in compiling and analyzing witness tree databases recorded from 1752–1899 in what is now the Monongahela National Forest (MNF) in eastern WV indicate a lower minimum potential elevation of red spruce historically (509 m) than previous models. The witness trees also revealed specific topographic position preferences of red spruce for north and northwest slope aspects and relative slope positions that vary by geomorphic context (Thomas-Van Gundy et al., 2012). Nauman et al. (2015) found that the spatial distribution of spodic soil properties is associated with the occurrence of red spruce and eastern hemlock witness trees, and follow similar topographic controls to those found in the analysis by Thomas-Van Gundy et al. (2012). Results from Nauman et al. (2015) also indicate that spodic soil properties are much more widespread in WV than previously thought, and concluded that this also likely represents a much greater extent of conifers historically. Estimations of pre-disturbance spatial distribution of red spruce might indicate historic affinity for topographically driven cool and moist microclimates that included the highest ridgelines, cooler slope aspects not in rain shadows, and narrow valleys that foster cold air drainage and foggy inversions (Nauman et al., 2015; Thomas-Van Gundy et al., 2012).

Red spruce appears to benefit greatly from its ectomycorrhizal (ECM) associated (Brundrett, 2009; Glenn et al., 1991) adaptations to acidic and nutrient poor settings (Blum et al., 2002), which also promotes SOC accumulation (Averill et al., 2014), and podzolization (van Breemen et al., 2000). Red spruce produces nutrient-poor litter (especially in Ca^{2+}) relative to other North American tree species (compare from: Berg and McClaugherty, 2008; Côté and Fyles, 1994; Friedland et al., 1988; Rustad and Fernandez, 1998). Eastern hemlock has also been shown to promote podzolization in a similar manner (Hix and Barnes, 1984). Hemlock is at the cooler edge of its range in this study area, and often times co-dominates with red spruce, but not always. So we hypothesized that red spruce and hemlock should promote podzolization and O horizon accumulation based on findings from previously discussed forest composition effects on soils (Herbauts and Buyl, 1981; Hix and Barnes, 1984; Lundström et al., 2000a; Miles, 1985; Ranger and Nys, 1994; Sauer et al., 2007; Sohet et al., 1988). We would also expect that the extensive areas of historic red spruce converted to non-ECM associated and more base cation promoting species like red maple (*Acer rubrum*) and black cherry (*Prunus serotina*) in WV have probably lost organic material from O horizons and B horizons (Averill et al., 2014; Barrett and Schaetzl, 1998; Brundrett, 2009; Comas and Eissenstat, 2009; Hix and Barnes, 1984; Hole, 1975; Miles, 1985). This was exacerbated by the large scale fires documented in WV after areas were clear-cut (Hopkins, 1899; Minckler, 1945; Pauley, 2008; Pielke, 1981). Despite the potential loss of much of the SOC pools we would expect in Spodosols associated with these stands, the Fe and Al sesquioxide accumulations in the subsurface soil are more stable and can persist as a 'memory' of the prior forest composition (Barrett and Schaetzl, 1998; Lundström et al., 2000b; Nauman et al., 2015; Parfitt, 2009).

Recent work related to ESD development in the MNF has suggested that spodic soil morphology in the MNF was linked to past red spruce and commonly associated eastern hemlock distributions (Nauman et al., 2015; Nowacki and Wendt, 2010; Teets, 2013). Nauman et al. (2015) were able to map this using spodic soil properties, which can be used in the delineation of the newly developed spodic shale uplands conifer forest (SCF) ecological site that this study will further document. They also showed that there was a positive linear relationship between the current relative conifer composition and the thickness of O horizons. We hypothesized that the areas of northern hardwood on the SCF ecological site are alternative ecological states that were converted from a spruce or spruce–hemlock dominated state by the railroad era timber harvest disturbance. We think this disturbance and conversion has resulted in large losses of O horizon material and associated C stocks, and so we build on the analysis of Nauman et al. (2015) to (i) link the relationship between O horizon depth and forest composition to ecological site state and transition models, (ii) spatialize both the SCF ecological site and the associated state and transition model using detailed current forest inventories, and (iii) estimate the potential C stocks in WV that could be accumulated in O horizons by restoring areas in alternative SCF ecological states to a spruce–hemlock dominated state according to the prescribed SCF ecological site description (Teets, 2013).

MATERIALS AND METHODS

Study Area

The study extended across the higher elevations of the Chemung and Hampshire geologic formations in parts of the MNF (Fig. 1). These are acid geologies primarily composed of shale and siltstone parent materials with minor inclusions of sandstone (WVGES, 1968). The area has a moist udic to perudic soil moisture regime, with annual precipitation ranging from 1118 to 1524 mm (44–60 inches; Arguez et al., 2014), varying by elevation and topographic rain shadow effects. Mean annual temperature ranges from 6 to 8.3°C (Arguez et al., 2014), varying by elevation, slope aspect, and cold air drainage patterns. The elevations of sites examined ranged from 880 to 1320 m,

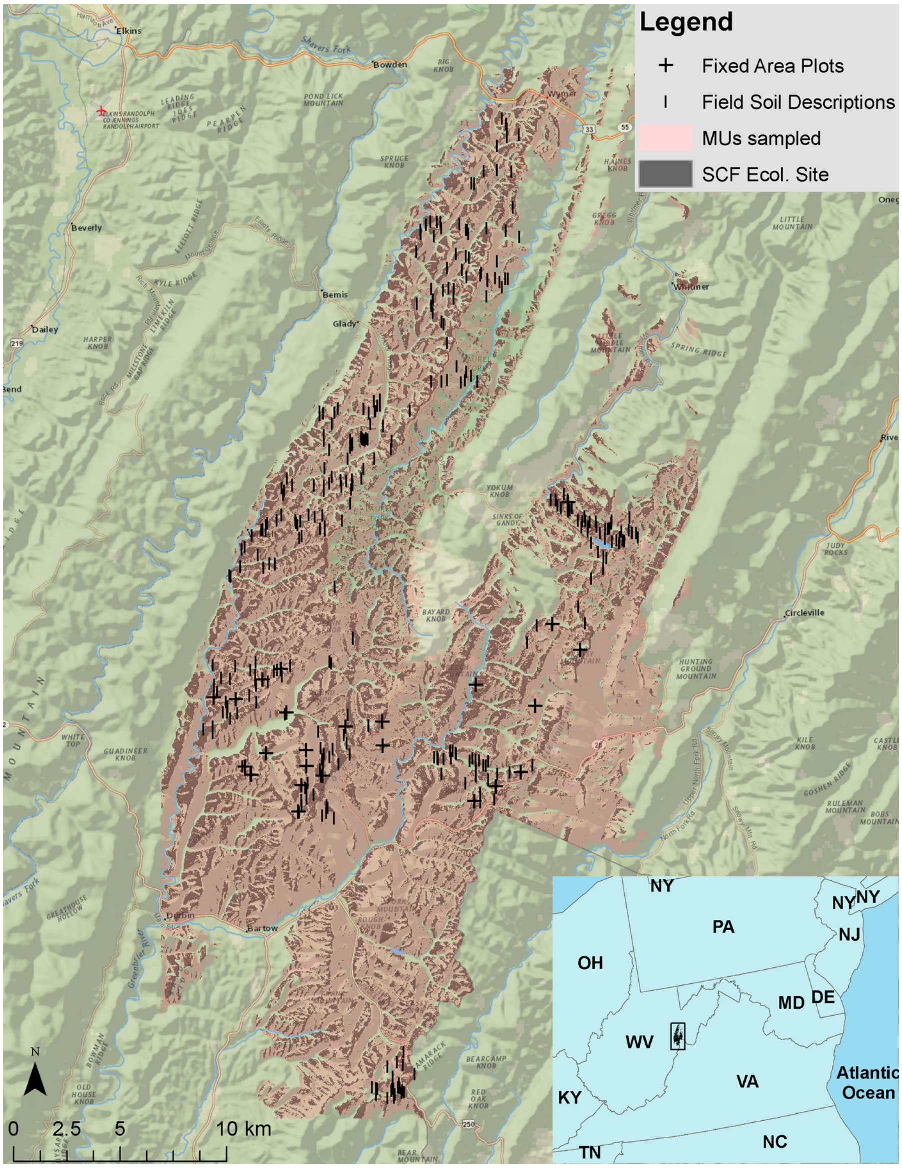

Fig. 1. Soil survey map units (MU) sampled for spodic probability model creation by Nauman et al. (2015), areas of the Spodic Shale Uplands Conifer Forest (SCF) ecological site, and data collection locations overlaid on ArcGIS 10 National Geographic mapping baselayer.

which spans the approximate boundary (~1100 m) between the mesic and frigid soil temperature regimes (Lietzke and McGuire, 1987; Stanley and Ciolkosz, 1981). The topography in the area includes flat narrow ridgetops, steep mountainsides, occasional rock outcrops, and deep and narrow river valleys. Within slopes there are benches, hollows, and nose slopes along with cradle-knoll micro-relief that mitigate how water, energy, and materials are distributed in the soil system.

Vegetation observed in these areas consists of northern hardwood and spruce-hemlock dominated stands as well as mixed composition stands where hardwood and spruce-hemlock co-dominate. Common tree species observed in the study area plots include red maple, sugar maple, mountain maple (*Acer spicatum*), striped maple (*Acer pensylvanicum*), red spruce, eastern hemlock, yellow birch (*Betula alleghaniensis*), sweet birch (*Betula lenta*), American basswood (*Tilia americana*), white ash (*Fraxinus americana*), northern red oak, black cherry, American beech (*Fagus grandifolia*), mountain magnolia (*Magnolia fraseri*), and cucumber magnolia (*Magnolia acuminata*). Commonly seen shrubs include mountain holly (*Ilex montana*), mountain laurel (*Kalmia latifolia*), and rhododendron (*Rhododendron spp.*), as well as shrubby root sprouts as a result of the beech bark disease complex (Shigo, 1972). Common herbaceous and ground cover species include New York fern (*Thelypteris noveboracensis*) intermediate woodfern (*Dryopteris intermedia*), hypnum moss (*Hypnum imponens*), liverwort (*Bazzania trilobata*), three *Lycopodium* species, *Viola* spp., and three *Carex* species.

Data Collection and Analysis

Three types of soils data were collected as part of this research: (i) extensive point observations of soil morphological properties (n = 322), (ii) detailed pedon descriptions at selected sites with associated comprehensive laboratory characterization of soil physical and chemical properties (n = 7), and (iii) fixed-area forest vegetation plots with detailed pedon descriptions and limited soil laboratory characterization data (n = 24). Data collected at all visited locations included detailed field descriptions of the soil morphology at hand-excavated pits with a focus on spodic morphology expression (i.e., spodic intensity; Table 1). Spodic intensity (SI) was determined on a zero (non-spodic) to two (well-expressed Spodosol) scale by 0.5 increments based on colors, horizon characteristics, and smeariness observations (See Table 1) typical of ‘spodic soil materials’ in U.S. *Soil Taxonomy* (Schoeneberger et al., 2002; Soil Survey Staff et al., 1999). Data were collected by a variety of local soil scientists associated mostly with the USDA-NRCS, USDA-Forest Service (FS), and West Virginia University (WVU). Soil descriptions were made consistent with U.S. national soil survey standards (Schoeneberger et al., 2002). Site locations were selected to evaluate soils derived from Devonian parent materials including mainly shale with sandstone inclusions on stable upland landscape positions for the purpose of soil survey update and preliminary ESD reconnaissance. Soil map units from the USDA-NRCS Soil Survey Geographic Database (SSURGO) (Soil Survey Staff et al., 2011) chosen for sampling included three locally common soil series: Mandy (Loamy-skeletal, mixed, active, frigid Spodic Dystrudepts; formerly a Typic Dystrudept), Berks (Loamy-skeletal, mixed, active, mesic Typic Dystrudepts), and Dekalb (Loamy-skeletal, siliceous, active, mesic Typic

Table 1. Description of spodic intensity (SI) classes based on observable field morphology.

SI class†	Description
0.0	No evidence of podzolization.
0.5	Very weak expression of podzolization. There is only slight physical evidence of podzolization. A slightly redder hue and higher value is present at the top of the B horizon, but the hue is less than one Munsell hue redder than an underlying horizon. The soil is non-smeary††.
1.0	Weak expression of podzolization (spodic intergrade, half the field soil profiles analyzed in the laboratory described as 1.0 keyed to Spodosols). Spodic materials are present, but they do not always meet the criteria for a diagnostic spodic horizon. A subtle Bs horizon is present. The Bs horizon is one Munsell hue redder than an underlying horizon. Bhs material is usually absent. An albic E horizon is not present. The spodic materials are sometimes weakly smeary.
1.5	Moderate expression of podzolization (Spodosol). Spodic materials are present as a diagnostic spodic horizon. A moderately expressed Bs horizon is present, often with pockets of Bhs material. An albic E horizon is not present. The spodic materials are often weakly smeary.
2.0	Strong expression of podzolization (well-expressed Spodosol). A diagnostic spodic horizon is present usually underlying an albic E horizon. A Bhs or Bh horizon is continuous across at least 85% of the pedon. The spodic materials are often moderately smeary.

† See Fig. 3 in Nauman et al. (2015) for laboratory analysis supporting SI field ratings.

†† Smeariness (Schoeneberger et al. [2002, p. 2–65]) is a physical observation about how moistened soil samples fail when they are squeezed and rubbed between the thumb and forefinger. Smeariness can help identify spodic soil materials.

Dystrudepts). Overstory and understory vegetation species lists were also noted at every location. Additional details about the sampling design and laboratory analysis can be found in Nauman et al. (2015).

Given the fire history in the study area, efforts were made to search for charcoal within the exposed soil profile and the four satellite O horizon observation points. When charcoal was found, the depth was noted and a representative sample was collected. In the lab, the size and shape of the charcoal pieces were recorded before sending them for ^{14}C analysis at the Northern Institute of Applied Climate Science (Michigan Tech. University, Houghton, MI) following the methods of Vogel et al. (1987), with a $\delta^{13}C$ correction applied to account for isotopic fractionation (Stuiver and Polach, 1977).

Fixed area forest plots employed for data collection were 20 by 20 m in shape and oriented with the slope aspect. Diameter at breast height (dbh) was measured on all trees >7 cm dbh. From measured dbh values and species tallies, importance (IMP) values (Eq. [2]; e.g., Adams et al., 2010) were calculated for red spruce and eastern hemlock.

$$\mathrm{IMP} = 0.5[(\frac{\text{species basal area}}{\text{plot tree basal area}}) + (\frac{\text{species count}}{\text{plot tree count}})] \qquad [2]$$

At plots, O horizon thicknesses were observed at the soil profile as well as at the center of each plot quadrant ($n = 5$ per plot). We added the importance of red spruce to that of hemlock to get a 'conifer importance' (CNIMP), which Nauman et al. (2015) showed to be a primary long-term 'organism' soil formation driver for podzolization, and shorter term controlling factor on forest floor thickness in these areas. We also summarized forest types at plots as 'conifer' (CNIMP > 0.75), 'mixed' (CNIMP 0.25–0.75), and 'hardwood' (CNIMP < 0.25) to help in plotting data.

Studies indicate that current conifer communities are much reduced compared with historic pre-disturbance conditions (e.g., Thomas-Van Gundy et al., 2012), and that current conifer relationship with SI is not as consistent as that with O horizon depth (Nauman et al., 2015). We contend that the Al and Fe accumulations reflected in SI visual cues and smeariness observations are longer lived signs of past vegetation than organic C. Thus, we assume that soils now under hardwood cover with spodic Al-Fe accumulations were under conifer before harvest and fire disturbance and had thicker O horizons in that previous state (Barrett and Schaetzl, 1998; Hix and Barnes, 1984; Lundström et al., 2000b; Parfitt, 2009). We suspect that O horizons have adjusted much more quickly to forest composition changes, and maintain closer correspondence to the current forest state.

Pedoecological Mapping and Restoration Carbon Sequestration Estimates

O horizon development in the MNF represents a potentially large pool of sequestered C. We estimated potential O horizon C accumulation based on our premises that (i) disturbance-driven forest conversion from conifer to hardwood will result in decreases in O horizon thickness, and (ii) that digital mapping methods will facilitate spatial representation of areas under different forest covers. Carbon accumulation estimates were based on the idea that returning areas with Spodosols under hardwood cover to the hypothesized historic reference conifer ecological state would increase O horizon thickness. This restoration to conifer could be accomplished by managing hardwood sites with spruce or hemlock in the understory with overhead release (Rentch et al., 2007, 2010) or under-planting with spruce and hemlock and later release if no recruitment is currently present. Figure 2 shows the state and transition model created for the spodic shale upland conifer forest (SCF) ecological site that represents the different forest compositions seen currently (Teets, 2013). We aimed to estimate the areas in the two logged states (Boxes 2 and 3 in Fig. 2) and estimate how much O horizon C would be added to areas in these states when restored to the conifer reference state (Box 1 in Fig. 2). To do this we combined analysis of the field point observations, detailed pedon data, plot data, and a forest inventory map (Byers et al., 2013) to map ecological states of the SCF and determine how much O horizon C could be restored by managing back to a reference state (Fig. 3).

Areas of SCF were estimated by choosing a spodic probability threshold value of 0.7 from the spatial model in Nauman et al. (2015), which maximized delineation of Spodosols with a rea-

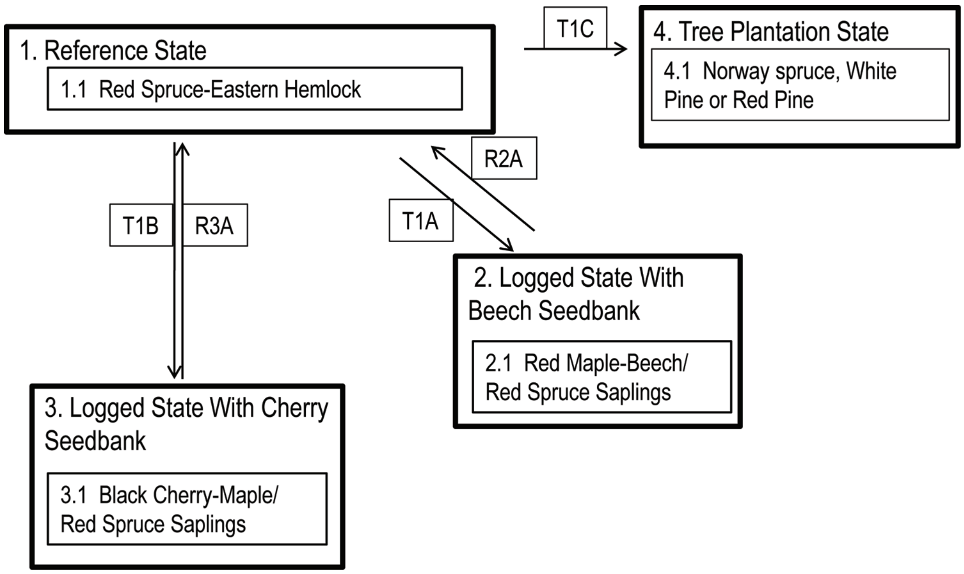

Fig. 2. State and transition model from spodic shale upland conifer forest (SCF) (Teets, 2013). T1C represents a transition to plantation by active planting after logging and/or fire. T1B and T1A are transitions to hardwood states by logging and/or fire and natural regeneration thereafter. Both R2A and R3A represent restoration pathways by overhead release of spruce and/or hemlock by selective harvest of hardwoods.

sonable degree of confidence (62–72% user accuracy from transect validation points and forest plot data). We also used forest plot data to see if current conifer dominated plots were within the 0.7 spodic probability cutoff to help verify this threshold. Then these areas within the SCF currently in a logged hardwood state (Boxes 2 and 3 in Fig. 2) or in transition from these states to the reference state were estimated in three ways. First, a current forest inventory (Byers et al., 2013) was overlaid with areas having spodic probabilities above the threshold to determine proportion of areas in a hardwood state or mixed conifer-hardwood composition (in transition between states). We present a map of the overlaid ecological states from the forest inventory and the SCF map as an example of a pedoecological map that provides field scale management prescription units for land managers based on a framework using digital soil mapping and ESD toolsets (Fig. 3).

The last two approaches for estimating areas in logged hardwood states only allowed for estimates of C sequestration to compare with that derived from the Byers et al. (2013) map, but not actual maps of the different ecological states. For the first of these, field observation sites above the 0.7 spodic probability threshold from the 1/3 withheld validation set in Nauman et al. (2015) were analyzed to see how many locations had no conifer in the forest overstory species to estimate what proportion of the study area was in a logged hardwood state. Third, the fixed-area forest vegetation plots determined to belong to the SCF in Fig. 2 were analyzed to

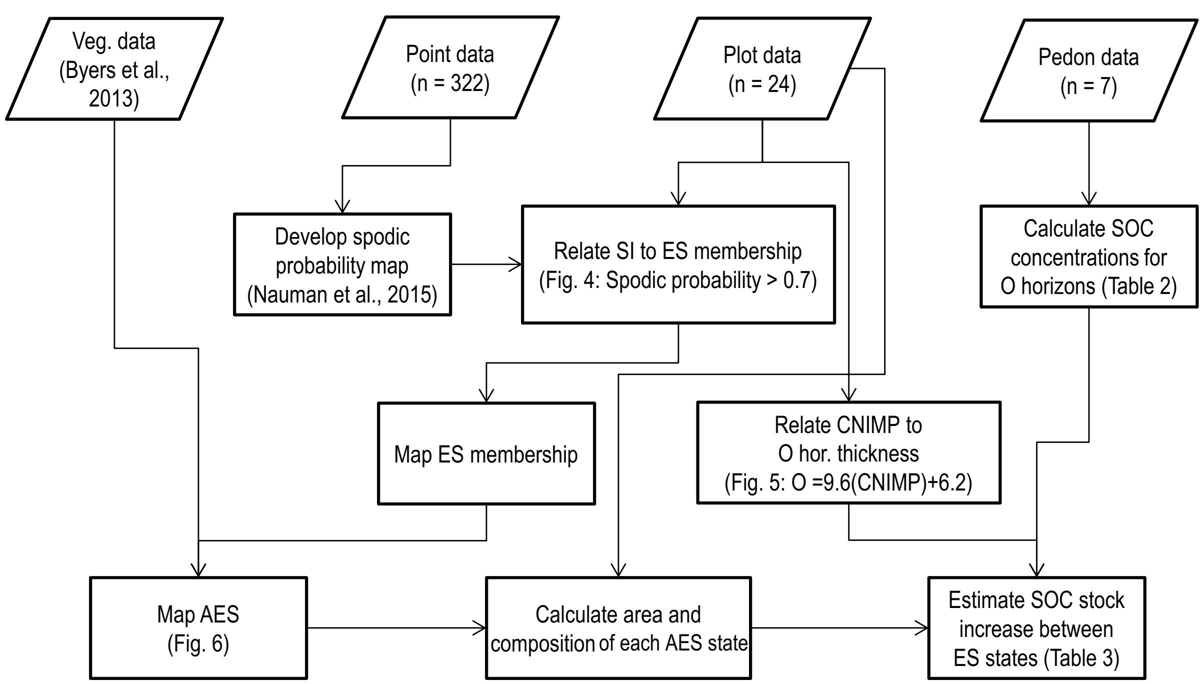

Fig. 3. Flow chart of data analysis used in pedoecological restoration estimates. Areal estimates of the spodic shale uplands conifer forest (SCF) ecological site (ES) extents were derived from point data spatial modeling of spodic intensity (SI). ES membership determination was made using forest plot data. Alternative ecological states (AESs) were determined primarily from Byers et al. (2013) overlaid on the spodic probability map. O horizon accumulation estimates were calculated by combining the relationship of conifer importance (CNIMP) to O horizon thickness with estimates of soil organic carbon (SOC) concentrations from laboratory data and geographical information system(GIS) maps of AESs. Forest composition of each AES was determined from attribution of mapping units from Byers et al. (2013) and refining specific CNIMP values in each polygon type from forest plot data collected in this study.

Table 2. Organic C calculations for O horizons from laboratory analysis used in restoration predictions of C sequestration.

Horizon	Average organic C	Average bulk density	Average % of O depth	g C cm^{-3}	Total weighted average in O horizons
	wt %	g cm^{-3}			g C cm^{-3}
Oi	47.5	0.063	20.6%	0.0301	
Oe	44.4	0.088	44.0%	0.0390	0.0572
Oa	43.1	0.222	35.4%	0.0958	

see what proportion fell into alternative logged hardwood states and areas that were in mixed-composition transitions to the reference state. The proportions found in these three methods were then multiplied by the mapped area above the spodic probability threshold in the map units sampled to determine potential restorable areas in those map units. We also scaled the proportions in the study area out to all areas in WV at elevations above the minimum elevation of our study (880 m) to make extrapolative estimates regarding how much C accumulation might result from restoration of spruce-hemlock in similar areas around the state. This extrapolative estimate is heavily weighted on assumptions that relationships and proportions are consistent outside of our study area in greater WV, but we feel it is a conservative estimate based on even thicker O horizons being associated with conifer states on the higher ridgelines in WV (T. Nauman, unpublished data, 2013), and also because Byers et al. (2013) show overall conifer composition proportions relatively consistent with those in our study area across their entire spatial estimate of historic WV red spruce extent.

Once potential restoration areas were identified, we used O horizon laboratory organic C (Method 4H2a; Soil Survey laboratory Staff et al., 2004) and frame bulk density (Method 3B5a; Soil Survey Laboratory Staff et al., 2004) estimates of the seven representative pedons sampled for laboratory analysis for calculating potential new C stocks. Total organic C estimates were averaged for Oi, Oe, and Oa horizons from soil profiles analyzed at the Kellog Soil Survey Laboratory (Table 2). Average proportions of Oi, Oe, and Oa in O horizons in the three reference state Spodosol profiles sampled were assumed as the proportions in new O horizon formation (Table 2). Total weighted average volumetric C (grams organic C per cubic centimeter) was used to estimate C additions from a given accrual of O horizon thickness over a certain area. Potential accrual of new O horizon depth was based on the slope of the linear relationship in forest plots between conifer importance and average plot O horizon depth.

We set restoration targets to the relative conifer (spruce + hemlock) basal area of 84.4% calculated by averaging the high and low listings in the ESD reference community basal area descriptions (Teets, 2013). Because relative basal area and conifer importance in our data were essentially the same (R^2 = 0.99, slope = 0.96) with the best agreement above 80% basal area, we translated the target to a conifer importance (CNIMP) of 84.4% because CNIMP had a better overall relationship with O horizons than relative basal area, although both were significantly correlated to O horizon thickness. It was unclear why CNIMP had a better relationship to O horizon, but it was possibly due to it incorporating both relative density and basal area into one factor representing forest composition. We assumed that to reach the restoration target, hardwood sites would need to increase in CNIMP by 76.3% because these sites averaged 8.1% in our forest plots that fell into the hardwood state as mapped by Byers et al. (2013). Because mixed transition plots averaged 53.1%, we used a 31.3% CNIMP increase for their target. Total areas of logged states and mixed transitions were multiplied by the estimated O horizon depth accruals for those states based on restoration targets and the slope of the O horizon-CNIMP relationship to get a total O horizon accumulation volume. The volume was then multiplied by the total weighted average of C density of O horizons (0.0572 g cm^{-3}; Table 2) to get a total mass of C predicted to be added to O horizons by meeting those restoration targets. The Oa/A horizons encountered in two laboratory profiles were assumed to be 66% Oa, and one A/Oa horizon recorded was assumed to be 33% Oa for calculating O horizon depth proportions, and bulk densities were scaled down by those factors as well. To adjust Oa C percentages in these same horizons, which are higher than A horizons in general, the C values were multiplied by 1.33 in Oa/A and 1.66 in A/Oa (educated guess), which in all cases produced C percentages slightly lower (conservative estimate) than the one uniquely measured Oa horizon C percentage of 47.3%.

RESULTS

Conifer Importance and O horizons

Conifer importances at SCF plots show positive correlation with the thickness of O horizons (Fig. 4). O horizon response to conifer importance was 0.96 cm of O horizon thickness increase per 10% of CNIMP expansion on average. It is important to note that CNIMP does not include any calculation of site productivity or herbaceous composition; it is solely based on the relative composition of red spruce and hemlock versus other tree species with dbh values greater than 7 cm. Therefore, this association is somewhat independent of site productivity and a range of other potential site variability. This relationship was chosen for restoration C sequestration calculations to model how changing species composition can affect O horizon C pools. This relationship was from plots with SI values of 1–2 (Spodosols or soils very nearly classifying as a Spodosol), where we think the reference spruce-hemlock dominated stands described by the SCF ecological site would have been most likely present historically based on the work of Nauman et al. (2015).

If we assume that Fig. 4 represents a compositional control on O horizon thickness, it still does not address the timeframe necessary for O horizon to adjust to forest composition changes. Fortunately, two of the forest plots we sampled were dense, even-aged red spruce stands (CNIMP = 86.3 and 100%) with charcoal evidence of burning after historic harvest. Breast height

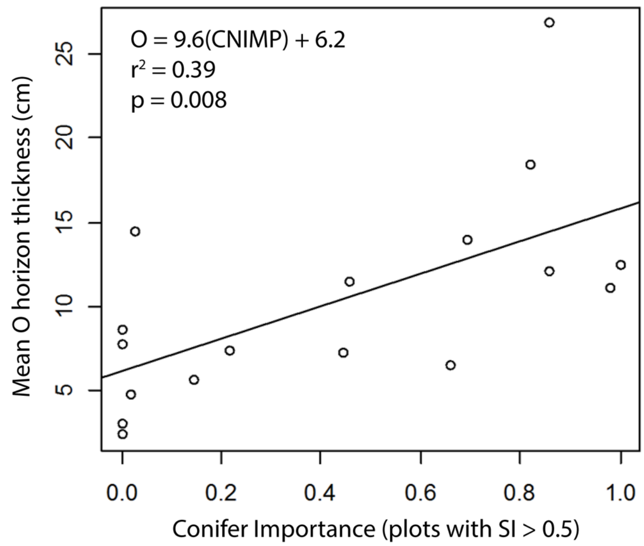

Fig. 4. Relationship between conifer importance (CNIMP) and O horizon depth at plots with weak to strong spodic properties observed (SI > 0.5, see Table 1).

tree cores of the three biggest spruce at both sites averaged 65 and 60 growth rings with a range of 52 to 70, suggesting stand ages of roughly 60 to 80 yr old. Abundant subangular charcoal was found at the interface between the O and E horizons at both sites, indicating that the O horizon had likely burned off before this cohort was established. Radiocarbon dates of the charcoal at these sites were 1808 ± 25 and 1923 ± 30 yr. Both sites had very similar O horizon thickness averages of 12.1 and 12.5 cm. This contrasts with O horizons at sites in the SCF that are currently under exclusively hardwood cover with similar or older ages (5.4 cm average). Interestingly, at the three conifer dominated plots older than 100 yr in average tree core counts (n = 3 per plot), the average O horizon thickness increased to 18.8 cm. At these older plots, we observed only one site with no charcoal evidence

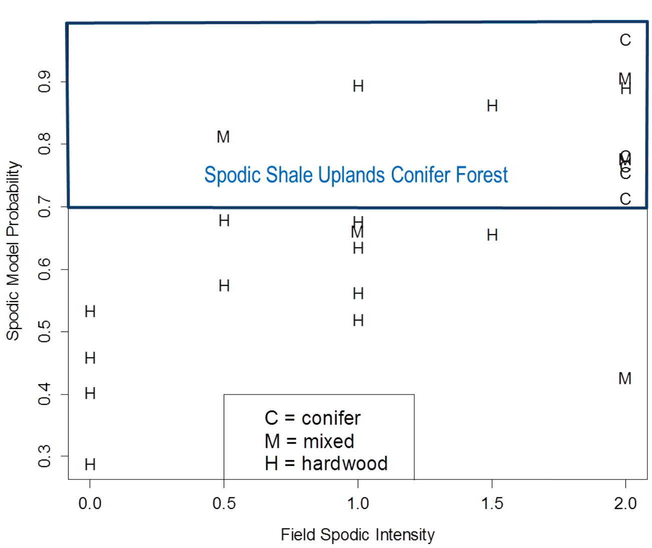

Fig. 5. Graph of fixed area plot spodic intensity (SI) values (*x* axis) versus probabilities from the random forest spodic probability model (*y* axis). Blue outlines delineate the spodic shale uplands conifer forest (SCF) ecological site, and letters represent dominant tree composition groups.

of past fire, and the average O horizon depth there was 26.8 cm with a maximum of 37 cm.

Plot Data

Plot data supported the choice of a 0.7 probability threshold as a basis for inclusion into the SCF ecological site with the vast majority of Spodosols and conifer dominated plots falling above that value (Fig. 5). The plot data shows a positive trend between observed SI and the predicted spodic probability. Most plots currently under conifer dominated and mixed hardwood-conifer also fall into the SCF. A few high mixed and hardwood outliers in SI categories of 0.5, 1.0, and 1.5 plotted above the 0.7 spodic probability threshold into the SCF ecological site. Another outlier in the SI 2.0 category fell below 0.7, out of the SCF, unexpectedly. However, laboratory Al and Fe ammonium oxalate extractions of the soil samples taken at these plots suggests that almost all of the field soil descriptions described as an SI ≥ 1.5 classify as Spodosols, and up to half of the profiles with an SI label of 1.0 would barely classify as Spodosols. This indicates that the high outliers are still mostly Spodosols, consistent with being grouped into the SCF ecological site, excepting the SI 0.5 category plot with a probability above the 0.7 threshold (Fig. 5). The low outlier plot (SI = 2.0) was examined and had an incorrect slope aspect value attributed to it by the geographical information system (GIS) model used in Nauman et al. (2015) when compared with the field observed aspect. Slope aspect was heavily weighted in the spatial modeling and likely caused an errant probability to be attributed.

Pedoecological Mapping and O horizon Carbon Accumulation

From the spodic probability threshold of 0.7 chosen to represent the SCF ecological site, a map was made to determine areas that could be potentially restored from logged alternative states to the reference spruce-hemlock conifer state (Fig. 6). The resulting map delineated 31% of the study area SSURGO map units that were originally sampled as SCF. A current forest cover map (Byers et al., 2013) was intersected with these areas to determine that 16.5% was mapped as conifer dominated, 73.6% as hardwood (or small patches of pasture), and 9.9% in mixed conifer-hardwood. Of the model validation field observations (withheld from model building) in Nauman et al. (2015) that fell into the SCF ecological site, 53.3% had no conifer species listed as present in the overstory, and were thus assumed to be in a hardwood state. Mixed states were not decipherable from these observations due to the qualitative nature of the vegetation data. Hardwood sites made up 36.4% of SCF fixed area forest plots, mixed sites made up 18.25% of plots, with the remaining plots being conifer. These different derivations of percentages of the ecological states (hardwood, mixed, conifer) were multiplied by the total area of the study area map units to get a range of estimates of the aerial extent of current stands in a hardwood state or mixed transition (Table 3).

Potential C sequestration estimates based on restoring to reference state conifer importance levels combined aerial estimates of

SCF stands in a hardwood state or mixed transition with expected O horizon accumulation (Table 3). We estimated that between 0.29 and 0.52 Tg of C would accumulate in the sampled study area soil map units. This approximation was extrapolated to all areas in WV with an elevation higher than 880 m (the minimum elevation of the study observations) by calculating the ratio of the greater WV area to the area of the study map units and multiplying our study area estimates by that ratio (12.73). Totals for the >880 m area ranged from 3.74 to 6.62 Tg (Table 3). These estimates are based on the calculation that, on average, areas in hardwood states would add 7.32 cm of O horizon material and mixed transition sites would add 3.0 cm based on the linear relationship between CNIMP and O horizon thickness in Fig. 4.

DISCUSSION

We build on the link between spodic morphology and historic reference spruce-hemlock communities (Nauman et al., 2015), to show how O horizons have likely changed since European colonization and railroad era timber harvest related disturbance. Early settler land clearing, industrial timber harvest and related fires, and resulting forest composition changes probably caused large losses of soil C stocks in the forest floor, which have somewhat returned in areas where spruce and hemlock have recolonized. Large fires were promoted immediately before and during the railroad era by a combination of hunters and ranchers slashing and burning forest, large scale timber harvest, the drying of O horizon due to timber removal, railroad ignition sources, and large pest outbreaks concurrent to harvest efforts leaving many dead standing spruce (Hopkins, 1899; Korstian, 1937; Minckler, 1945; Pauley, 2008; Pielke, 1981). However, despite this prolific historic disturbance, results also indicate potential for red spruce restoration, which could result in large accumulations of O horizon material.

It is important to temper the interpretation of our results with the fact that considerable error exists in our data. The spodic probability maps had an error rate of ~30%, the regression of conifer importance and O horizon depth only explains 39% of the variance, and there is error associated with the laboratory data. These sources of error should be taken into account when planning operations. We recommend having trained soil scientists and forest ecologists involved in restoration projects to check soil and vegetation at potential restoration locations to ensure that the proper ecological site is being targeted for management operations.

Timing of O horizon Accumulation

Charcoal data at two even aged red spruce sites was examined to help understand potential O horizon accumulation rates. Both stands were 65- to 80-yr old based on tree ring counts of three dominant trees at each site. Both sites also had very similar O horizon thicknesses of approximately 12 cm accumulated on top of nearly continuous charcoal layers at the O-E horizon boundaries. Based on the modern dates of the charcoal, it seems reasonable that most of these O horizons have accumulated within the last century within the development of these dense red spruce stands. The ^{14}C dates (1808 ± 25 and 1923 ± 30) indicate a possible connection to historic post-logging accounts of long lasting fires in the area (Hopkins, 1899; Korstian, 1937; Minckler, 1945; Pauley, 2008; Pielke, 1981). These dates support modern fires that we postulate followed forest clearing for pasture (1808) and railroad era timber harvest disturbance (1923).

The O horizon depths at the radiocarbon dated sites match well with the conifer importance relationships with O horizon depth where a conifer dominated stand with a high CNIMP would be predicted to have an O horizon near 15 cm thick (Fig. 4). Since the two even aged stands are not as old (~70 yr old)

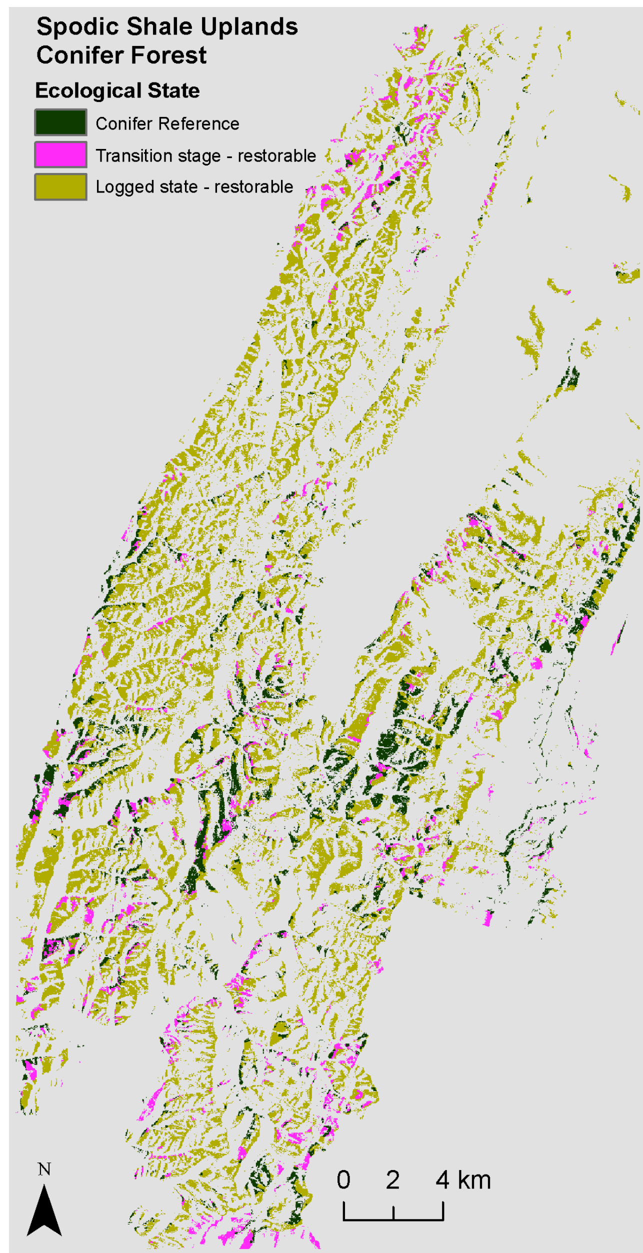

Fig. 6. Pedoecological map of ecological states and state transitions within areas delineated as the spodic shale uplands conifer forest (SCF) ecological site using the 70% probability threshold. Differing colors represent the various states and transitional areas in SCF derived from overlaying the Byers et al. (2013) forest inventory map onto the spodic probability map.

Table 3. Organic C accumulation calculations for spodic shale uplands conifer forest (SCF) ecological site restoration scenarios. Estimates of total organic C stored in O horizons above 880 m elevation in WV, assumed the same areal proportions of of spodic soils (31%) and alternative ecological states within spodic areas (e.g. 9.9% mixed and 73.6% hardwood from Byers et al., 2013). With the assumption that these were consistent across these elevations, the ratio (12.73) of the total area in WV above 880m (645,438 ha) to the study area extent (50,715 ha) was used as a multiplier to scale up C sequestration estimates. O horizon accumulations are based on estimates of conifer importances (CNIMP) from study plots within the different mapped states and their differences from the reference state (CNIMP of 0.844). The regression slope from Fig. 4 (0.96 cm O horizon per 0.1 increase in CNIMP) was used to determine O horizon accumulation for scenario of restoring to the reference state.

Ecological state areal estimate source	Hardwood state	Mixed transition	Total C seq. in study map units	Total C seq. in WV above 880 m††
	ha		Tg†	
Byers et al. (2013)	11758	1581	0.52	6.62
1/3 validation set	8513	n/a	0.36	4.54
2013 plots	5814	2907	0.29	3.74
Conifer IMP deficit est.	0.76	0.31		
O horizon increase, cm	7.32	3.00		

† Teragrams = 10^{12} g.

†† Scaled up by 12.73 factor from study map unit values based on assumptions of similar proportions of spodic soils and alternative ecological states across these elevations.

as some of the other red spruce dominated plots we sampled, it make sense that they have only accumulated 12 cm of O horizon, but with time we think they would likely reach 15 cm and perhaps even greater thicknesses. Therefore, restoration and perpetuation of red spruce can potentially facilitate significant O horizon buildup within a century and beyond, especially when compared with the O horizon depths found at similarly aged hardwood stands (5.4 cm average at our plots). This general timeframe is similar to that observed by Schaetzl (1994) in O horizon buildup after fire in northern hardwoods. Schaetzl (1994) showed a very similar amount of O horizon accumulation (~5.5 cm) within the first century after fire in northern hardwoods, and diminishing rates of accumulation thereafter.

If we think beyond a century, our observations indicate even deeper O horizons to be possible (18.8 cm average at plots with red spruce > 100 yr in age, $n = 3$ plots). Of these older plots, the only location with no charcoal found had considerably deeper O horizons (26.8 cm average with a 37 cm maximum). This could be representative of an old growth reference condition that is consistent with the reporting of deep O horizons in more undisturbed red spruce stands (Byers et al., 2010 Korstian, 1937; Minckler, 1945; Pauley, 2008; Pielke, 1981).

We acknowledge that there are alternative interpretations of these dating results, and are uncertain as to how spruce at these two plots with ^{14}C charcoal dates were able to regenerate so dominantly after what appeared to have been an intense fire based on the nearly continuous layer of charcoal found at the O-E horizon interfaces at these plots. Our first impression of these plots was that they were planted because of very even and somewhat linear spacing of the trees, similar to Norway spruce stands planted in the area, but no records of red spruce plantations exist in the area during that time period to our knowledge. It should also be noted that the timeframe for development of these pure spruce plots (and underlying O horizons) is faster (~100 yr) than the timeframe needed to restore spruce in the more common restoration scenario where areas in a hardwood state have spruce in the understory that can be released by selected harvest. More reasonable timeframes of 200+ yr are likely for release-based restoration as presented in Rentch et al. (2010). However, nearby higher elevation areas like Cheat Mountain were observed to have thick monoculture spruce thickets recruiting in many areas that will likely produce similar pure spruce stands, and thus similar forest floor accumulation rates and should be considered in C balance projections for those areas.

Implications of Red Spruce Restoration for Wildlife and Climate Change

Different studies have indicated that red spruce will mostly disappear from the central Appalachians within the century under even the best climate change scenarios (Butler et al., 2015; Hansen et al., 2001; Iverson et al., 2008; Prasad et al., 2007; Young et al., 2010) implying that debate regarding the benefits of red spruce restoration is moot. However, considerable uncertainty is associated with these type of projections because of the difficulty incorporating dispersal rates, biodiversity dynamics, and unforeseen scenarios (Hansen et al., 2001). Studies of the red spruce–northern hardwood ecotone in New England have often focused on the elevation of the transition and the associated ecological changes (Siccama, 1974; Beckage et al., 2008). Late twentieth century decreases in the growth of red spruce and upward shifts of the ecotone have largely been attributed to climate warming, but cannot rule out pollution and competition as co-factors (Beckage et al., 2008; McLaughlin et al., 1987). Hamburg and Cogbill (1988) were able to show that climate was probably more influential than air pollution (e.g., acid rain) in red spruce decline since 1800. Earlier work on the spruce-hardwood ecotone in Vermont also showed a correspondence between more acidic soils with thicker forest floors and red spruce dominated areas, but didn't report as much specificity between spruce and spodic properties (Siccama, 1974; Young, 1934). However, modern studies must account for the possibility that the vast harvest disturbance of forests associated with European colonization has favored hardwood incursion into formerly conifer influenced areas (Nowacki et al., 2010; Pielke, 1981) that might be reflected in soils with spodic properties currently under hardwood cover.

Our results do suggest red spruce restoration could play a role in climate change mitigation by helping sequester C into soils during new stand development, and it is possible that establishing red spruce canopy might also keep soil temperatures lower (Hobbie et al., 2006). It might be difficult to discern the effects of climate change on red spruce range because so much of the northeastern USA has been intensively disturbed since the industrial revolution. Indeed both the Monongahela National Forest witness tree database (Thomas-Van Gundy et al., 2012) and historical accounts (Hopkins, 1899) indicate that the red spruce range stretched much lower in elevation (500–700 m) in certain topographies than current distributions would indicate. Several other recent studies show that red spruce populations are actually recovering and expanding (Nowacki et al., 2010; Rollins et al., 2010). Red spruce restoration may also become increasingly important where it co-dominates with eastern hemlock due to the projected loss of hemlock to the hemlock woolly adelgid (*Adelges tsugae*) (Hessl and Pederson, 2013).

Tree cover and species also have an effect on climate dynamics and soil temperatures. Data from Hobbie et al. (2006) showed that spruce and fir species were associated with lower mean annual soil temperatures and decreased litter decomposition in a common garden experiment with a variety of tree species. Pielke (1981) reported on how the vast destruction of forests circa 1900 increased regional temperature, which then began to lower again around 1940 with the return of the mostly hardwood forest. It is also unclear what role land use conversion plays in climate change by affecting surface radiative dynamics (Pielke, 2001, 2005; Pielke et al., 2002). The potential effect of forest mitigation of surface warming should be further investigated with respect to historically native conifer communities, and might uncover further resilience of red spruce and similar communities against warming temperatures.

Restoring hardwood areas of the SCF ecological site to the proposed reference conifer state will also potentially produce significant habitat for rare species in addition to C sequestration benefits. The endangered Cheat Mountain salamander (*Plethodon nettingi*) has been associated with red spruce forest communities in parts of our spodic probability model footprint and could benefit from restoration (Dillard et al., 2008a, 2008b; Pauley, 2008). The formerly endangered Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*) has also been linked to forests with influential red spruce components (Menzel et al., 2004; Menzel et al., 2006a, 2006b; Odom et al., 2001) and would likely benefit from restoration efforts.

Potential C sequestration calculations associated with ESD restoration scenarios (Table 3) for just the areas in alternative ecological states within study area map units sampled (Fig. 6) represent the C equivalent (0.52 Tg) of consuming 4.4 million barrels of oil according to the EPA C equivalents calculator (United States Environmental Protection Agency, 2014). This amounts to about the same amount of C as 23% of the 18.9 million barrels of oil used in USA each day (United States Environmental Protection Agency, 2014). Since much of the rest of WV above the minimum elevation of this study (880 m) is a very similar landscape to the sampled study areas, we decided to extrapolate these results based on the assumption that a similar proportion of those areas would be Spodosols soils (31%). We also looked at the Byers et al. (2013) map to justify that the ratio of conifer to hardwood ecological states is also similar across these areas (9.9% mixed and 73.6% hardwood in our study area versus 66% hardwood and 17% mixed across entire Byers et al. (2013) map). Based on these assumptions, C sequestration calculations were scaled up to all the potential areas of Spodosols in alternative hardwood ecological states in WV above an elevation of 880 m by simply multiplying the estimates by a factor of the ratio of the extent of WV above 880 m (645,438 ha) to the area of the study map units sampled (50,715 ha) to get a value of 12.73. The extrapolation amounted to a value of 6.62 Tg of C (Table 3), the equivalent of 56.4 million barrels of oil, or about 3 d of oil consumption in the USA (United States Environmental Protection Agency, 2014).

The O horizon C sequestration calculations represent a conservative estimate of potential C storage that would likely result from restoration. Subsurface C storage potential is not included in these calculations, and is a significant pool in Spodosols that can respond quickly to disturbance (Barrett and Schaetzl, 1998; Hix and Barnes, 1984; Hole, 1975, 1976). Based on some concurrent sampling on Cheat Mountain, WV, we also think that conifer composition in higher and wetter areas in WV, relative to our study area, might have twice the impact on O horizon accumulation (i.e., ~2.1 cm O horizon thickness increase per 10% increase in conifer importance, $r^2 = 70$, $p = 0.0001$, T. Nauman, unpublished data, 2013).

Although our estimates in WV account for a small portion of global emissions, it is indicative of how temperate forest encroachment into cooler subalpine and boreal conifer systems is a significant contributor to atmospheric CO_2 through combinations of human disturbance, as seen in this study, and climate change (e.g., Hamburg and Cogbill, 1988). Lal (2005) showed that the boreal and tundra systems represent significantly larger organic C pools than temperate forests, and that they are potentially the most vulnerable to climate change. Lal (2005) also points out that as much as two thirds of global forest C stocks are in soil organic C, and that in boreal systems this ratio is even higher.

CONCLUSIONS

Disturbance induced changes in high elevation forests of WV likely resulted in large historic losses in soil organic C stocks, and represent a sizeable potential for C sequestration. Carbon pools appear to respond relatively quickly to the contraction or expansion of subalpine conifer species like red spruce and eastern hemlock. Spodic soil properties help reflect the extent of the conifer forest composition before the vast ecological disturbance wrought on the land by European colonization and railroad-driven timber harvest in the central Appalachians. Key to telling this story is considering the time scale required for differing soil processes to react to changes in environment, and choosing a framework to help model those changes. In this case we used a state and transition model from an ecological site description.

Although climate change is a daunting challenge and species like red spruce seem to be ill-fated from some perspectives, they also might represent a significant mitigation potential as new data emerges. Alternatively, if red spruce is lost, similar species that promote podzolization and C accumulation including other selected *Tsuga, Larix, Picea*, *Pinus*, and *Abies* species could serve as alternatives. What seems pretty clear is that losing this forest community will result in increased C emissions from soil pools, and the potential loss of rare and sensitive wildlife species and pools of biodiversity. Restoration of red spruce and similar species represents one of many potential climate change mitigation and ecological restoration options that society will need to evaluate in our efforts to achieve global sustainability.

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