Forest Ecology and Management 306 (2013) 216–225

Contents lists available at SciVerse ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem

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ARTICLE INFO

Article history:
Received 27 September 2012
Received in revised form 20 June 2013
Accepted 21 June 2013

Keywords:
Resilience
Climate change
Adaptive management
Forest disturbance recovery
Carbon sequestration
Modeling

ABSTRACT

Ecosystems and ecosystem services are subjected to both typical disturbances (e.g., fire) and shifting climatic baselines resulting from anthropogenic drivers. Recovery from these perturbations is of prime interest to researchers and land managers. We explore how differing regeneration of the coniferous forest to specific disturbances and a shifting climate are mediated through managerial responses, in terms of both species composition and an important ecosystem service, carbon sequestration in the southern Rocky Mountains, Colorado, USA. 112 sites across a variety of disturbance histories were surveyed for post-fire regeneration; carbon stock growth was then simulated in the US Forest Service Forest Vegetation Simulator under a variety of climate change scenarios for 100 years. Simultaneously, we simulated three managerial responses to the disturbance: no action, planting of local species (resilience-oriented management), and planting of the most climatically suitable species (adaptation-oriented management). These managerial responses simulate varying levels of intervention which attempt to maintain forest properties and associated carbon stocks. Carbon stocks, initially, were more resilient than the coniferous forest system; areas with little coniferous regeneration recovered carbon at a similar pace due to an influx of deciduous seedlings. However, future climate exerts a strong influence on carbon stocks. Both the no-action scenario and the resilience-oriented management scenario transitioned to non-forest by the end of the simulation period, due to climatic changes. Active, adaptation-oriented management, which included establishment of non-local species, maintained forest structure and carbon stocks under most future climate projections, albeit at lower densities. So while this preserves the presence of a forest, it does not preserve the presence of a specific forest. However, for ecosystem services associated with the mere existence of forest cover (e.g., carbon stocks and general forest habitat), this may be sufficient. In a sense, disturbances are opportunities for more climatically-adapted species/communities to establish, although the complexities of assisted migration and novel ecosystems remain.

1. Introduction

Ecosystems, and the services they provide, will experience two types of perturbations in the future: discrete disturbances such as fire and the slower change imposed by shifting climatic regimes. Disturbances are inevitable in the majority of forests around the world. Continuation of a forest in any given location through multiple cycles of disturbances is contingent upon ecosystem resilience: the recovery of the system to a similar state (Holling, 1973; Gunderson, 2000). This may be rapid, depending on initial post-disturbance establishment (Brown and Johnstone, 2012) or protracted through early and late successional stages. In the future, however, recovery will take place in an era of changing temperatures, precipitation, and disturbance regimes, and so any long-term projection of ecosystem recovery must take those factors into account. Disturbances are expected to increase across a wide range of forest ecosystems (e.g., Dale et al., 2001; Flannigan et al., 2009) and may trigger shifts in species ranges (e.g., Johnstone and Chapin, 2003) or eliminate forests all together (Brown and Johnstone, 2012). Interactions between multiple disturbances may cause novel disturbance characteristics (Buma and Wessman, 2011), differential recovery (D'Amato et al., 2011; Brown and Johnstone, 2012), and/or regime shifts (Paine et al., 1998). Given the potential for disturbances to cause such large changes in ecosystem character, and the likely increasing rates of disturbance, it is important to investigate their impact on ecosystems and their properties going forward.

Carbon storage in ecosystems is related to local climate (Davidson and Janssens, 2006), topography, species composition and structure (Wessman et al., 2004), soil characteristics (Lal, 2005) and disturbance history (Brown and Johnstone, 2011), among other factors. Forest growth sequesters a large amount of carbon in biomass and soils (1086 Pg globally, Lal, 2005). They do not store that carbon in perpetuity, however; disturbances and mortality return a portion of that carbon to the atmosphere through either combustion (direct

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0378-1127/$ - see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.foreco.2013.06.044

carbon emissions, usually as CO_2) or through the resultant decomposition of the killed biomass. Yet if the forest recovers to a similar structure and density, the total carbon exchange will be neutral over the time period of recovery (Kashian et al., 2006). This recovery may be fairly rapid; regenerating vegetation may quickly take up enough carbon to offset decomposition. Investigation in Canadian lodgepole pine forests with heavy insect infestations found that even high mortality stands were a growing season carbon sink within a few years (Brown et al., 2010). This was attributed to the understory vegetation rapidly fixing carbon in response to the newly available resources freed up by the death of the overstory trees (Bowler et al., 2012). Given that the system recovers to a similar state, it is likely that carbon stocks will recover as well. Changing ecosystem states, however, may have large impacts on total carbon stocks due to fundamental changes in plant structure, density, and soil inputs. For example, woody plant encroachment may have a strong effect on the carbon balance of the landscape, mainly through the increase in plant biomass (Wessman et al., 2004), although the magnitude of change depends upon moisture and other factors. In other cases, regimes may change without a concurrent shift in carbon stocks.

Using a forest growth simulator supported by extensive field measurements in disturbed forest landscapes, carbon stock recovery was simulated in the context of a changing climate and various regeneration/management scenarios. The pre-disturbance ecosystem was mature spruce-fir forest; the post-disturbance recovery is highly heterogeneous – in some areas, dominated by coniferous regeneration, in others, deciduous or graminoid, as a result of the different disturbance histories in the area (Buma and Wessman, 2011). Here the resilience of the coniferous forest is defined as the relative amount of conifer regeneration one decade post-fire. This heterogeneity is followed through the coming century and total ecosystem carbon (non-mineral soil) was simulated to determine the relative influence of disturbance history/regeneration, management, and climate change on forest carbon stocks. We ask the following questions:

1. How does differing post-fire regeneration (both in species and amount) influence carbon stock growth under various climate scenarios?
2. How do differing management scenarios affect carbon stock growth under various climate scenarios?

2. Methods

2.1. Site and plot design

A combination of disturbances (blowdown, logging, fire) in the Routt National Forest in north central Colorado, USA, resulted in a spectrum of post-fire recovery rates and trajectories in a subalpine spruce-fir forest (Buma and Wessman 2011, 2012). The forest (approx. 2700–3300 m elevation) is comprised of spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), lodgepole pine (*Pinus contorta*), and quaking aspen (*Populus tremuloides*). Precipitation averages approximately 1000 mm a year, mostly as snow (NRCS 2010). The blowdown (October 1997, approx. 10,000 ha, Baker et al., 2002) left a mosaic of severities, from zero to near 100% blowdown. Salvage logging, conducted post-blowdown (1998–2001), took place on approximately 900 ha. In the summer of 2002, the Mt. Zirkel fire complex burned approximately 12,500 ha of undisturbed, blown-down, and salvage logged forest. The blowdown altered fire severity (Kulakowski and Veblen, 2007), as well as postfire recovery (Buma and Wessman, 2012).

112 15 $\times$ 15 m plots were censused for post-fire regeneration of woody plant species and woody debris. All plots were located in areas identified as closed-canopy spruce-fir forests prior to the disturbances (US Forest Service RIS (Resource Information System) data and personal observation, unpublished). These plots were grouped according to their disturbance history (Fig. 1): no/low blowdown and fire (0–20 downed trees/ha; $n = 27$), medium blowdown/fire (20–55 downed trees/ha; $n = 41$), areas of high blowdown/fire (55 + downed trees/ha; $n = 33$), and logged blowdown/fire (55 + downed trees/ha and prior logging; $n = 11$). Standing dead trees (snags) were measured on a subset of plots (no/low = 22 plots; medium = 8 plots; high = 10 plots; logged = 8 plots). This grouping scheme corresponds with decreasing conifer resilience (as defined by seedling densities, Fig. 1 middle); areas with little to no conifer regeneration were considered non-resilient, for example.

At each plot, seedlings were counted and measured for height. Coarse woody debris (CWD) totals were estimated via methods from Brown (1974). All standing dead trees were measured for their diameter at breast height (DBH) and height. To compare recovering carbon stocks to undisturbed forests and estimate belowground carbon stocks, 10 additional plots were established within undisturbed spruce-fir forests. The same measurements were conducted, with species, DBH and height recorded for all trees.

2.2. FVS

Carbon dynamics were simulated in the USDA Forest Vegetation Simulator (FVS) using the carbon sub-model contained in the Fires and Fuels extension (FVS-FFE, Rebain, 2010) and the climate extension module (Climate-FVS, Crookston et al., 2010), for 100 years (2010–2109). FVS is a well-known forest simulator often used for carbon and disturbance simulation (e.g., Hurteau and North, 2009) and is parameterized for different geographical regions; the Central Rockies variant was used here. Plant growth (calculated decadally) occurs based on species-specific relationships between local climate/topography and the local community (e.g., crown closure and tree density), and was calibrated according to DBH (diameter at breast height) and height allometric relationships (Jenkins et al., 2003). Mortality occurs via two processes, background mortality (species and size specific probabilities) and density dependent mortality, which is species specific and determined based on stand density and species shade tolerance. Regeneration is user specified, and so was implemented according to the management strategies described below. Each plot was grown independently. Elevation was obtained from the national elevation dataset (USGS 2009), with a resolution of 30 m. Aspect and slope were calculated from this dataset using ArcMap (ESRI, 2010).

Live seedlings were input into the simulator and their total C was calculated according to Jenkins et al. (2003). Field estimated CWD was used to initialize downed debris loads; FVS-FFE allometrics were used to calculate C in initial snags (based on field survey). Because species could not be determined for the burned snags, all snags were considered Engelmann spruce, the dominant species in unburned stands. Forest floor (e.g., duff) and shrub/herb layers were calculated using Smith and Heath (2002) via FVS-FFE and based on canopy cover percent, age, and dominant tree species.

The initial amount of dead coarse roots could not be determined for each plot because it was impossible to determine pre-fire tree sizes and densities with any certainty. In addition, logging removed the majority of the tree boles, and some stumps, so their coarse roots would be unaccounted for if stumps or snags were used to initialize belowground dead coarse roots. Instead, the mean belowground coarse root carbon totals (live and dead) from the undisturbed plots was used to initialize all the burned plots. This makes the assumption that the burned plots were compositionally similar to the control plots, and is a statistically conservative decision, reducing the variability between the treatments. The