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Communication

Accountable Accounting: Carbon-Based Management on Marginal Lands

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Abstract: Substantial discussion exists concerning the best land use options for mitigating greenhouse gas (GHG) emissions on marginal land. Emissions-mitigating land use options include displacement of fossil fuels via biofuel production and afforestation. Comparing C recovery dynamics under these different options is crucial to assessing the efficacy of offset programs. In this paper, we focus on forest recovery on marginal land, and show that there is substantial inaccuracy and discrepancy in the literature concerning carbon accumulation. We find that uncertainty in carbon accumulation occurs in estimations of carbon stocks and models of carbon dynamics over time. We suggest that analyses to date have been largely unsuccessful at determining reliable trends in site recovery due to broad land use categories, a failure to consider the effect of current and post-restoration management, and problems with meta-analysis. Understanding of C recovery could be greatly improved with increased data collection on pre-restoration site quality, prior land use history, and management practices as well as increased methodological standardization. Finally, given the current and likely future uncertainty in C dynamics, we recommend carbon mitigation potential should not be the only environmental service driving land use decisions on marginal lands.

Keywords: forest carbon sequestration; carbon dynamics; land-use change; forest recovery; afforestation

1. Introduction

There is a growing interest in using marginal land for greenhouse gas (GHG) mitigation. Recent estimates have suggested that 1107–1141 million hectares of marginal land may exist, of which 385–472 million hectares is thought to be abandoned agricultural land [1,2]. GHG mitigation options currently under consideration for marginal land (defined below) include on-site carbon (C) sequestration via reforestation or afforestation, as well as fossil fuel displacement via the production of bioenergy crops [3]. The relative efficacy of these different options is likely to be highly context-dependent, and thus determining the optimal GHG mitigation strategy for any given location requires a thorough understanding of carbon dynamics.

However, there are several key gaps and limitations in the existing literature on carbon dynamics that prevents the precise evaluation of these mitigation strategies, particularly with regards to marginal land. These include inadequacies in C pool quantification, broad categorization of land use types and conversion histories, inadequate measurement and/or reporting of critical site-specific factors, and the inevitable uncertainty associated with future predictions [4,5]. These inaccuracies limit the efficacy of global and national policies aimed at reducing atmospheric carbon levels through terrestrial mitigation strategies [6]. Without reliable input data to support site-specific C sequestration potential, resources may be invested in sequestration or offset projects that are either (a) unlikely to achieve their stated objectives or (b) sub-optimal compared to other land use options for C offsets (5).

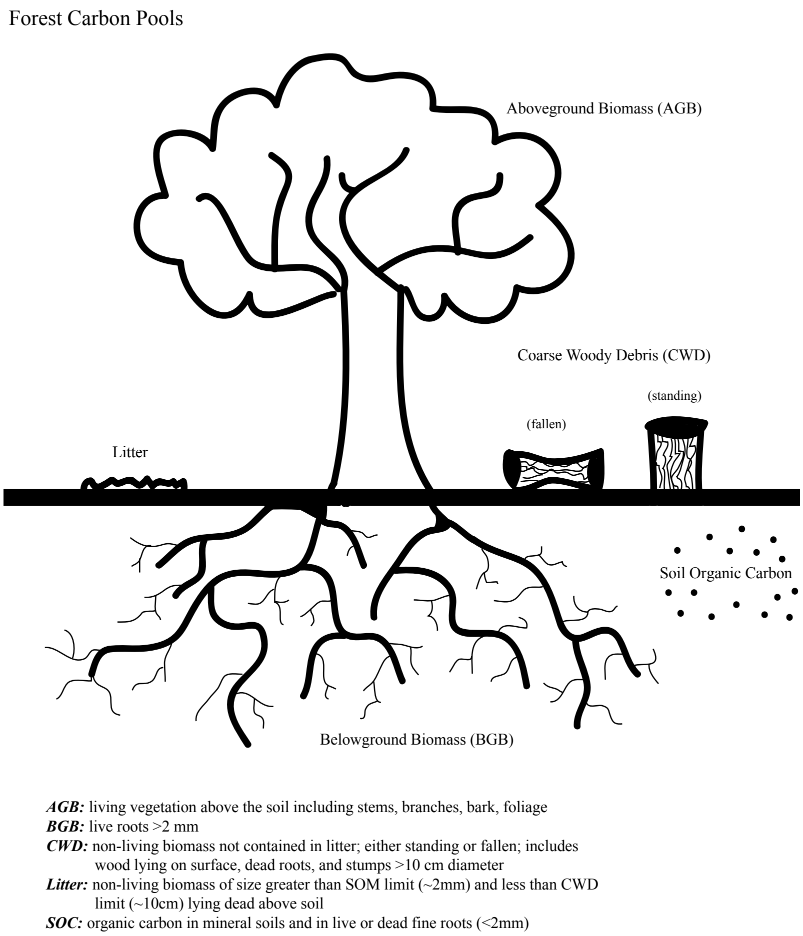
The objective of this paper is to synthesize trends and challenges in quantifying C on recovering marginal lands. We focus in particular on the C sequestration potential of recovering forests, either via natural succession or afforestation. We do not directly consider C offsets of bioenergy crops as this has been discussed in detail elsewhere (e.g., [7,8]). In the context of this paper, our use of the term marginal land refers specifically to land that: (a) is not being used for any clear economic or subsistence purposes; (b) has been substantially altered by humans from its original condition and is currently of limited value in terms of biodiversity and ecosystem services; and (c) has low potential yields based on soil, temperature, precipitation, elevation, and slope [9]. While the scope of our work is limited to the ecological aspects of C sequestration, it is essential to recognize that many social, legal, and ethical aspects of marginal land management still require detailed analysis and consideration.

The paper is organized as follows. We first summarize the difficulties of quantifying forest C stocks, noting key sources of inaccuracy and suggesting improvements in methodology and standardization. We then discuss barriers to the effective prediction of C recovery dynamics on marginal lands, and outline key data to be recorded and considered prior to any land management decision. We conclude with recommendations for future research and suggestions for policies to mitigate the potential pitfalls of carbon-based management schemes.

2. Quantification of Forest Carbon Stocks

A methodological understanding of C stock quantification is integral to investigating current discrepancies in the C recovery literature [10]. Forest carbon stocks are typically divided into five pools: aboveground biomass (AGB), belowground biomass (BGB), soil organic carbon (SOC), coarse woody debris (CWD), and litter (Figure 1) [11]. The proportions of C in each pool are affected by numerous factors, including forest age, which is highly relevant because different pools exhibit different rates of recovery following natural succession or afforestation [12]. The AGB and BGB pools increase during forest recovery, but the relative contribution of each pool in relation to regeneration method remains poorly understood due to a paucity of BGB studies [13,14]. Within the soil pool, C typically decreases shortly after restoration then increases to levels that equal [15–19] or exceed pre-restoration C stocks [20–23], but see exceptions observed in early succession: [16,24,25]. The initial decrease in soil C is due to a lag in leaf litter accumulation, which upon decay contributes C to the soil pool [26,27].

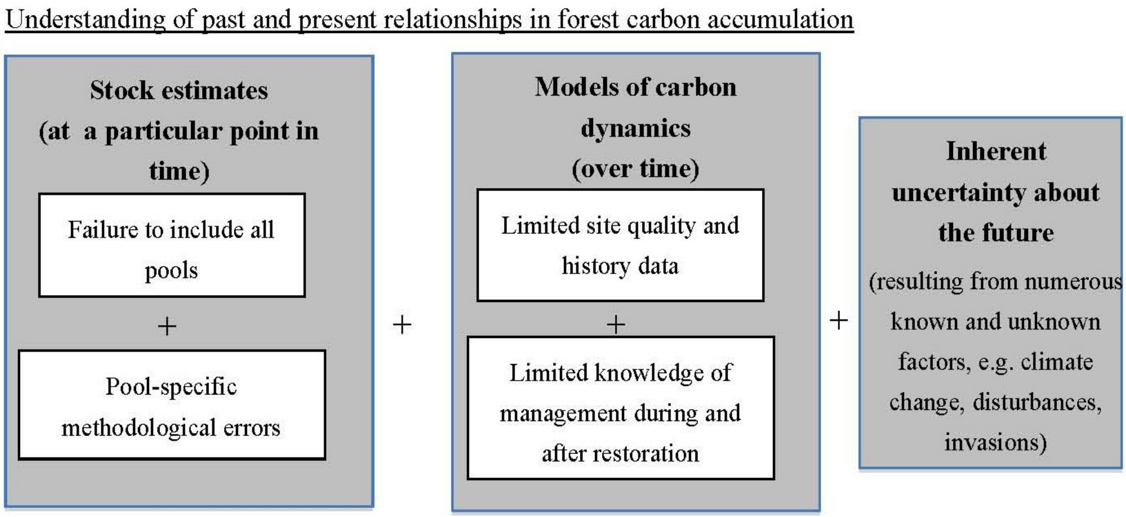
Figure 1. Carbon (C) continuously cycles between the atmosphere, the dead and living biomass, and soils of forest ecosystems [28]. For quantification and subsequent sampling purposes, it is recommended that forest C should be divided into five different pools: aboveground biomass, belowground biomass, soil, coarse woody debris, and litter [26]. The division of a forest ecosystem into C pools is shown below.



Sources: Sierra *et al.* [28], Dierkes [29], Ngo *et al.* [6].

Despite general knowledge of coarse trends such as these, the dynamic processes by which C cycles and accumulates remains poorly understood in forests [4,22,26], particularly regenerating forests [30], and precise stock proportions for any given forest can rarely be estimated with confidence. Sources of uncertainty in estimates of forest carbon sequestration occur at multiple temporal levels resulting from the interplay of numerous factors (Figure 2). The two main sources of uncertainty with regard to stock estimates are (1) failure to measure all C pools and (2) pool-specific methodological problems [28,31]. Below we discuss in detail the implications of these shortcomings for carbon centric land use decisions.

Figure 2. Sources of uncertainty in estimates of forest carbon sequestration. Uncertainty in forest carbon estimates occurs at multiple temporal levels for various reasons outlined below and discussed in the following sections.



2.1. Failure to Include All C Pools

Failure to include all five pools of C in studies gives an incomplete measurement of C accumulation during land restoration and makes it difficult to compare C accumulation across studies [32]. Most studies that quantify C stocks following land-use change solely include the aboveground biomass pool, with the inherent assumption that this is indicative of total C recovery [13]. However, solely looking at the AGB pool can lead to an overestimation of C stocks in earlier years of succession, failing to account for C losses in soil and debris respiration. There is also evidence that the BGB pool could be 68% larger than previously thought, and that allocation of biomass above *versus* belowground differs with regeneration method [33]. For example, Cuevas *et al.* [14] found that 44% of biomass production in a regenerating secondary forest was belowground *versus* only 6% in a paired afforested monoculture. While meta-analyses often ignore the BGB pool because of a paucity of data (e.g., [12]), including the BGB pool could drastically change results especially with regards to the sequestration potential of one regeneration method over another [13]. Similarly, studying changes in the SOC pool without accompanied litter C data is misleading because litter C decays into SOC pools. Failure to include the litter pool has led studies to conclude losses in SOC with regeneration, when in fact the net ground C remained the same due to increases in litter C from tree growth [22].

2.2. Pool-Specific Methodological Errors

In addition to inaccuracies arising from pool inclusion, there are many inaccuracies when quantifying the C in each individual pool. Estimations of C in the aboveground biomass (AGB) pool, where the majority of C sequestration during restoration occurs, vary significantly depending on methodology [10,34]. The most accurate method of calculating AGB C content is to apply *site-specific* allometric equations (derived from local destructive sampling) to forest inventory data [35]. The choice of allometric biomass equation can significantly affect the estimation of AGB C stocks, causing potentially large errors in AGB C quantification when inappropriate allometric equations are used [4,34,36]. A further source of error is the assumption that all trees contain a C content that is 50% its AGB, which is widely used despite little chemical verification [37]. Not only do most trees contain less than 50% C by weight, but studies are also finding significant variation in C content between species (e.g., Elias and Potvin [38] found a significant variation in tree C content of 32 tropical trees, ranging from 44.4% to 49.4%). Accounting for the variation in species-specific C content can reduce errors in estimates of AGB C sequestration by 10% in some cases [38], but this parameter is rarely considered in C estimations [37]. Additionally, several studies have found a significant difference in the C content of the heartwood *versus* sapwood of a tree, which should also be considered when processing wood core samples and calculating AGB C [39].

Belowground, the spatial heterogeneity of root biomass and consequent sampling challenges make the belowground biomass pool the most difficult to quantify [40]. When the pool is not ignored altogether, many studies use inaccurate root: shoot ratios (estimating BGB from AGB data) or inadequately sample their study sites due to time and financial constraints [41,42]. Inadequate sampling methodologies have recently been calculated to underestimate the global belowground biomass pool by 50%–68% [33,43]. Further errors occur from the loss of sample mass during preparation and storage. These differences in mass alone can underestimate root content by 1/3 [42]. While more accurate methods of belowground biomass sampling use soil monoliths and trench sampling, an adequate sample volume has yet to be determined [42,44]. Since SOC accumulates at different rates within different soil depths during site regeneration [20], the most accurate sampling methodology is to repetitively sample soil at 1–2 m depth at the same site. However, most studies sample only to much shallower depths often due to sampling difficulties in the degraded soils characteristic of marginal landscapes [4]. This data gap in deep soil C dynamics in conjunction with a paucity of studies on later successional SOC limits our understanding of soil nutrient stocks and dynamics [24].

2.3. The Importance of Standardized and Scientifically Sound Methodology

Before comparing C accumulation between sites, it is vital to understand the calculation of those C values and their level of reliability and accuracy. Above, we have demonstrated the multiple levels of uncertainty and variability in forest C estimations. It is not our intent to design extensive methods for accurate forest C quantification, as has been done extensively elsewhere [45–47]. Instead we stress the rationale behind these protocols, outlining the risks and errors incurred from falling short on the recommended sampling protocols. In doing so, we emphasize the importance in the global application of these standardized methods that is necessary to accurately determine trends in C recovery on marginal land [12,48]. Understanding the sources of uncertainty in C estimations allows decision makers to analyze whether or not a forest C study is accountable for a C-trading mechanism. Ideally, previously collected data should be reanalyzed when possible using the types of methodology presented by IPCC or the Center for Tropical Forest Science to provide a more complete global picture of C sequestration from land-use change. Without such standardization, it is difficult to reliably estimate C sequestration following land-use change [12].

3. Barriers to Effective Prediction of C Recovery Dynamics

Beyond the difficulties of quantifying existing C stocks at a particular point in time, further uncertainty arises when assessing the sequestration potential of a parcel of marginal land. Factors affecting C recovery (stocks and rates in all pools) include regeneration method, previous land use history, climate, geomorphology, latitude, tree species, age, and distance to seed source, among others [12,21,49–51]. However, there is little cohesion in the literature as to which parameter is most indicative of site recovery. From a critical review of individual studies and meta-analyses, we have determined that the literature is inconclusive about which biophysical parameters are most predictive of site recovery for three main reasons: (1) the use of broad categories of land use history; (2) a failure to consider the effect of current and post-restoration management; (3) problems with meta-analysis.

3.1. Broad Categories of Land Use History

Rather than collecting or finding detailed information on site quality, studies often consider land use history as a proxy for regeneration and production capacity. Here, site quality is defined as the ability of a piece of land to support vegetative growth and is influenced by a host of factors ranging from climate to soil structure. Previous work has demonstrated that land use category *i.e.*, cropland, pasture, or mining, duration of land use, and the number of types of previous land use are all indicative of pre-recovery site quality [12,17,52–56]. However, while similar land use histories often create similar site characteristics, leading to similar biomass accumulation trends [25], there are several problems with this approach.

First, most research fails to acknowledge that there is substantial variation of practices within all of the land use categories typically specified in the literature, *i.e.*, pasture, cleared, agriculture, fire [12,13]. For instance, within the category of agriculture, there exist a variety of practices ranging from swidden cultivation to intensively managed and fertilized cropland. Fertilized cropland usually has less degradation and higher fertility than lands under swidden cultivation, leading to higher biomass accumulation rates than on the ash-fertilized acidic soils following swidden cultivation [12,57]. While both types of landscapes are grouped into the category of abandoned agricultural land, it is clear that different management practices can lead to large differences in site quality [25]. This is also the case for the pasture land use category, where the details of pastoral activity (e.g., the duration of fallow periods, grazing intensity, and number of cycles) differentially impact the ability of abandoned pasture to regenerate forest [58,59]. Whereas old pastures are typically carbon-depleted and therefore experience greater rates of C sequestration into the soil pool than do younger pastures [60], the rate of biomass accumulation is greater on younger pastures [58,59].

Grouping all pastoral landscapes into one category for analysis overlooks these land use details that may be more indicative of regeneration potential. Letcher and Chazdon [53] warn of such "crude" division of land use categories. It is also important to consider that the distribution of land uses may reflect preferential site selection for desirable characteristics, confounding land use effects with actual site quality effects [61].

Despite these issues, several general trends emerge from the literature that can inform carbon-centric management practices, outlined in Table 1.

Table 1. Prior land use practices and their associated effect on forest recovery. These are trends generally agreed upon in the literature concerning C sequestration on abandoned land given certain site history characteristics.

Prior Land Use Characteristics	Effect on Forest Recovery
Fertilized > Not Fertilized	Fertilized cropland will likely experience higher rates of biomass accumulation than unfertilized land, due to increased soil nutrient availability.
Cropland > Pasture	Pasture lands tend to be more degraded than cropland, as low fertility and soil compaction cause reduced aeration and soil biological activity. However, this trend could be due to the preferential selection of less productive sites for pasture in the first place.
Young Pasture > Old Pasture	Younger pastures are less degraded than older pastures since they have not experienced as much soil compaction and nutrient loss from pastoral activity. Therefore, there is faster biomass accumulation on younger pastures than older pastures.
Longer fallow > Shorter fallow	Although an ideal fallow period to maintain site fertility has yet to be established in the literature, longer fallow periods allow the replenishment of site nutrients. Increased site nutrients are associated with increased biomass growth and subsequent C sequestration.
Fewer cycles > More cycles	Fewer cycles of cultivation means a less degraded landscape. A less degraded landscape has better site quality with the nutrients necessary for higher carbon accumulation rates.

Sources: Vesterdal and Rosenqvist [25], Kotto-same *et al.* [57], Hughes *et al.* [59], Milakovsky *et al.* [40], Klanderud *et al.* [62], Silver *et al.* [12], Fearnside and Guimares [63], Feldpausch [58].

Broad land use categories are not informative with regard to most of the factors discussed in Table 1. Thus, it is extremely important that before any land use decisions are made, a comprehensive assessment of biophysical site quality be undertaken [60]. Unfortunately, the majority of studies rarely report site quality data, which partially explains why the literature has been inconclusive in providing reliable predictors of site regeneration capacity [13]. On a final note, if forest regeneration is to proceed naturally without planting, it is also important to consider the surrounding landscape matrix and proximity to forest [64]. While the duration and intensity of previous land use influences within-site propagule availability, the surrounding landscape determines the seed arrival that is necessary for forest establishment [51].

3.2. Effect of Current and Post-Restoration Management

In the same way that site quality caused by land use history affects regeneration potential, land management during and after restoration affects C sequestration potential [65,66]. The frequent categorization of regenerating landscapes into the dichotomy of secondary forest or monoculture plantation often fails to consider the specific details of regeneration schemes implemented on marginal landscapes. Regeneration schemes vary widely in degrees of management intervention, such as thinning, fertilization, the planting of N-fixing species, mycorrhizal inoculation, and environmental restoration plantings [67–70]. These forms of assisted regeneration, with often extensive management interventions, can lead to higher C sequestration rates than those of unassisted stands [5,17]. For example, while it is generally believed that plantations sequester more C than secondary forests, Bonner *et al.* [13] found that this result disappears when one compares unfertilized plantations to secondary forests. Considering the effects of active management techniques on C accumulation rates is therefore important to avoid the confounding of these actions with pre-restoration site recovery potential.

3.3. Problems with Meta-Analysis

As noted in the introduction, much C policy today is being guided by the results of reviews and meta-analyses. The failure of most primary studies to report details on site quality, land use history, and management practices seriously limits the inferences that can be drawn from subsequent syntheses of these studies. For instance, the recent meta-analysis by Bonner *et al.* [13] is limited by a paucity of replicates for several important combinations of factors. While they found that tropical AGB accumulation was not significantly affected by previous land use, annual precipitation, or soil order, this is probably more a reflection of sample size limitations in meta-analyses than reality; evidence for this assertion can be found in related studies with different methodologies that have found these factors to significantly affect forest growth (e.g., [12,17,71]). Given the current state of the literature, even if a meta-analysis were to determine accurate global averages, these global estimates are unlikely to be applicable to any given individual site considering the variety of mechanisms that affect C sequestration potential and realization [4].

4. Research, Policy, and Decision-Making Implications

4.1. Research Implications

A thorough understanding of site carbon sequestration potential requires pairing carbon accumulation measurements with detailed knowledge of pre-restoration biophysical site characteristics. Accurate projections of future dynamics on marginal land will depend on detailed models that include most or all of the key factors discussed above. Thus, future studies should strive to provide detailed site quality data [60], as well as detailed land use history information (although it must be recognized that even extremely detailed land use history data is only coarsely indicative of pre-restoration biophysical attributes). In addition, because pre-restoration site quality interacts with subsequent management, these management actions should be investigated and reported. From our review of the literature, we have compiled a table of parameters deemed significant to assessing site C sequestration potential. This list combines the comparative parameters used in multiple studies and meta-analyses into one complete table. Given sites with similar geography and climate, the following data presented in Table 2 is necessary to assess the carbon sequestration potential of one site over another.

Table 2. Data necessary to assess a site's carbon sequestration potential. These data should ideally be collected in all sequestration studies and at sites prior to the implementation of any carbon land management scheme.

Data Category	Relevant Variables
Site Quality	A range of physical (slope, aspect, altitude) and soil characteristics including pH, bulk density, N, Ca, P, K, micronutrient levels, soil texture and structure, stone content *(pre-restoration data)*
Land Use History	Type, duration, number of cycles, number of types, fertilization, remnant vegetation, other detailed practices *(pre-restoration data)*
Management	Fertilizer applications during planting/natural succession, ploughing, tilling, mycorrhizal inoculations, thinning, planting of N-fixing species, any other strategies for assisted regeneration *(data during restoration)*

Sources: Feldpausch [58], Hughes *et al.* [59], Gamboa *et al.* [72], Silver *et al.* [12], Bonner *et al.* [13], Kasel and Bennett [55], Peichl *et al.* [65], Paul *et al.* [17], Holl [64].

Ideally, site quality data should be collected directly (*i.e.*, via soil testing), and management actions should be tracked and recorded as they occur. However, this of course may not possible if research begins after the recovery process has started. Potential ways to gather these important data include increased use of archival data and interviews with local inhabitants, as well as increased use of remotely sensed data [60]. However, it should be noted that while interviews with locals can fill important gaps in knowledge, interviews can be time consuming and provide information of varying quality [24]. In addition, while the use of aerial imaging will provide some aspects of land use history (e.g., [19]), it also leaves out many important site quality details such as fertilization regimes and intrinsic soil properties.

4.2. Policy and Decision-Making Implications

While the quantification of existing forest C stocks is error prone due to insufficient methodology and the failure to include all C pools, studies on *future* forest C recovery contain even more uncertainty because they involve *future* projections. Ziegler *et al.*'s [73] meta-analysis of 250 studies in Southeast Asia concludes that it is "virtually impossible" with our current state of knowledge to predict how land-use changes affect total ecosystem C stocks. Uncertainty at local scales and within individual pools may have little consequence for global C estimates, provided there is no systematic bias. However, even if this is true, these broad estimates have little relevance to any individual site, and it is at this local scale that land use decisions are actualized. A wide range of factors influences local C stocks and dynamics (e.g., climate, tree species, latitude, precipitation, proximity to forest), but these relationships have been poorly quantified to date [4]. As such, reliable C recovery projections for a particular parcel of land under a particular management regimen will likely remain elusive for some time to come.

Given these limitations, it is important for policy and decision makers to acknowledge these uncertainties when making decisions about the future use of marginal lands. Policy and decision makers would be better served by making land use decision in a broader context that acknowledges climate mitigation as one of the many environmental services forest landscapes may provide. For example, there are many other services provided by forests such as watershed protection, pollination, and biodiversity [58,74] and while the most carbon-positive land use transition may not maximize these non-carbon benefits [73], focusing on non-carbon benefits as well as carbon benefits may lead to more resilient systems. Illustrative examples include Thompson *et al.* [75] finding that monocultures are less resilient than a system containing a greater species richness, making a less C rich forest more favorable in the face of climate change and disturbance. Further, while afforested monocultures may be credited with more C sequestration [12], they are often more susceptible to disease and therefore result in less resilient ecosystems [76].

5. Conclusions

In conclusion, C recovery would be better understood with increased data collection on pre-restoration site quality, prior land use history, and management practices as well as increased methodological standardization. In addition, given the current and likely future uncertainty in C dynamics, we recommend carbon mitigation potential should not be the only environmental service driving land use decisions on marginal lands.

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Author Contributions

Tara L. DiRocco performed the research for the paper and wrote the first draft of the paper. Matthew D. Potts and Benjamin S. Ramage conceived the research framework and questions. Samuel G. Evans advised on the agricultural portion of the manuscript. All co-authors assisted with writing and revising successive drafts.

Conflicts of Interest

The authors declare no conflict of interest.

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