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Soil carbon stock changes due to edge effects in central Amazon forest fragments

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ABSTRACT

Amazon forest stocks large quantities of carbon both in plant biomass and in soil. Deforestation has accelerated the process of forest fragmentation in the Brazilian Amazon, resulting in changes in carbon stocks in both biomass and soil. Logging, including that under legal forest management, can create edge-like conditions inside the forest. We investigated the relationship between changes in carbon stocks in the soil and the distance to the nearest edge in forest remnants after about 30 years of isolation. We assessed the effect of edges using geographically weighted regression (GWR), which considers the non-stationary character of soil carbon stocks and assigns relative weights to the observations according to the distance between them. Data from 265 georeferenced plots distributed over 28 ha of forest fragments in the Manaus region were included in these analyses. Soil-carbon stocks were estimated for areas before (1984–1986) and after (2012–2013) isolation of the fragments. The GWR model indicated an apparent relationship between change in carbon stocks and distance from the edge ($R^2 = 0.79$). The largest changes occurred in plots located closest to the edges. In 202 plots $\leqslant$100 m from an edge, soil-carbon stock increased significantly ($p = 0.01$) by a mean of 1.34 Mg ha^{-1} over the $\sim$30-year period. Such changes in soil carbon stocks appear to be associated with higher rates of tree mortality caused by microclimatic changes in these areas. Increased necromass inputs combined with changes in composition and structure of vegetation may result in increased rates of decomposition of organic matter, transferring carbon to the soil compartment and increasing soil carbon stocks. Considering both "hard" edges adjacent to deforestation and "soft" edges in logging areas, the soil-carbon increase we measured implies an absorption of 6×10^6 MgC in Brazilian Amazonia. In hard edges maintained for $\sim$30 years, the soil-carbon increase offsets 8.3% of the carbon losses from "biomass collapse" in the first 100 m from a clearing. Soil carbon did not change significantly in 63 forest-interior plots, suggesting that global climate change has not yet had a detectible effect on this forest carbon compartment.

1. Introduction

The Amazon forest stocks large quantities of carbon in plant biomass (Nogueira et al., 2008, 2015; Saatchi et al., 2011) and in soil (Batjes, 2005; Batjes and Dijkshoorn, 1999; Fearnside, 2016; Fearnside and Barbosa, 1998). In the context of global warming these forests can play a strategic role in climate regulation (Fearnside, 1997). However, cumulative deforestation by 2015 (Brazil INPE, 2015) had destroyed 19.5% of Brazil's Amazonian forests. Annual deforestation rates declined from 2004 to 2012 and fluctuated around the 2012 level through July 2014. However, 2015 was marked by a rise in deforestation (Fearnside, 2015; Fonseca et al., 2015).

As a result of this process, continuous native forest cover has been replaced by a landscape dominated by isolated forest remnants in a matrix of farmland and pasture (Laurance and Bierregaard, 1997; Murcia, 1995; Saunders et al., 1991). The edge effect caused by fragmentation leads to increased tree mortality (Laurance et al., 1998) probably as a result of higher temperatures and decreased soil moisture at the forest edges compared to the forest interior (Camargo and Kapos, 1995), greater exposure to harsh winds (Rankin-de-Merona and Hutchings, 2001) and increased liana biomass at the forest edges (Laurance et al., 2014a,b).

Logging can produce edge-like conditions inside the forest because canopy gaps create a hotter and dryer microclimate and because inputs of necromass are increased from logging slash

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and logging-induced mortality (e.g., Asner et al., 2006; Broadbent et al., 2008). Canopy gaps and consequent microclimate alteration persist for 4–6 years after harvest (e.g., Gerwing, 2002), but frequent intrusion of fire and other disturbances means that many logged areas in Amazonia enter a cycle of continued degradation (e.g., Berenguer et al., 2014). Reduced impact logging (RIL) can reduce damage (Sist and Ferreira, 2007), and RIL normally has less canopy opening than the "conventional" logging that continues to be a common practice in Brazilian Amazonia (depending on the state, 46–65% of logged area is unlicensed: Monteiro et al., 2013; Silgueiro et al., 2015; a significant part of what is licensed is non-compliant with management requirements: e.g., Britto, 2015). Even RIL can result in up to 25% canopy opening (Jackson et al., 2002). Higher necromass inputs persist for over a decade even when increased mortality ceases shortly after the initial harvest (Blanc et al., 2009; Palace et al., 2007).

Economic activities in Amazonia that entail deforestation and forest degradation contribute substantial amounts of net emissions of greenhouse gases (Fearnside, 2000a). Estimates of emissions from conversion of forests into pastures do not explicitly consider soil-carbon stock changes in forest edges (Fearnside and Barbosa, 1998; Fearnside et al., 2009).

Deforestation in Brazilian Amazonia creates a landscape that is a mosaic of forest fragments embedded in a matrix of other land uses (mainly cattle pasture). The edges of these fragments lose substantial amounts of carbon from "biomass collapse" (Laurance et al., 1997). This carbon loss increases the impact of deforestation on global warming beyond the impact of carbon emission from the deforested areas themselves. However, the additional emission from edge formation only applies to the increase in the total length of edges in the region each year, not to the carbon loss from the much larger extent of edges present in the region that remains in place from each year to the next (Fearnside, 2000b). This is because the great majority of deforestation in Amazonia occurs by expansion of existing clearings into the surrounding forest, rather than by appearance of new clearings away from previously cleared areas. When existing clearings expand into adjacent forest, the forest edges are being cleared and the carbon stock in these areas has therefore already been reduced by the "biomass collapse" phenomenon. Counting the emission of deforestation based on the biomass of intact forest therefore would double-count the same carbon if the biomass-collapse emission has also been counted. The same reasoning that applies to biomass carbon stock changes also applies to soil carbon stock changes. Where Amazon forest is converted to cattle pasture (the predominant land use in deforested areas), soil carbon is lost under the normal system of pasture management (Fearnside and Barbosa, 1998).

The contribution of forest fragmentation to the balance of greenhouse-gas emissions is still poorly known. Most studies have focused on the evaluation of effects on plant biomass (Nascimento and Laurance, 2004, 2006) and litter (Didham, 1998; Sizer et al., 2000; Vasconcelos and Laurance, 2005; Vasconcelos and Luizão, 2004). Long-term effects of forest fragmentation on the stock of soil carbon remain unknown.

Amazonian soils store approximately 276 Mg of carbon per hectare at a depth of 0–8 m (Fearnside, 2016). Changes in forest structure that influence microclimate will affect the production and decomposition of organic matter, resulting in losses or gains of carbon stocks. The present study aims to assess changes in soil-carbon stocks due to edge effects in forest fragments that have been isolated for nearly 30 years.

2. Materials and methods

2.1. Study area

Our study was conducted on an experimentally fragmented landscape maintained by the Biological Dynamics of Forest Fragments Project (BDFFP). This project emerged during the discussions on the planning of protected areas known by the acronym "SLOSS" (Single Large Or Several Small reserves of equal area), which sought to assess the importance of the size of reserves for species conservation (Laurance et al., 2011). Forest fragments of different sizes (1, 10 and 100 ha) were isolated in three large cattle ranches for deployment of large-scale experiments in the early 1980s. The BDFFP's main objective was to establish the basis for assessments of the environmental consequences of deforestation and fragmentation on the Amazon rainforest. Isolated reserves were surrounded by cattle pastures.

Our study area is a 1000-km^2 experimental landscape that includes primary rainforest, forest fragments, and a matrix of cattle

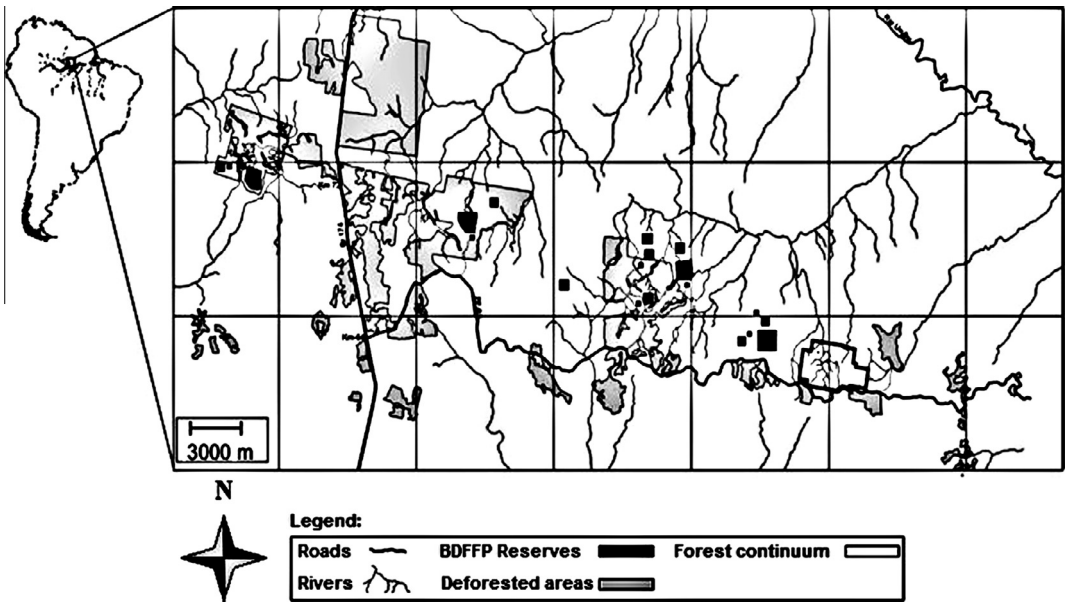

Fig. 1. Biological Dynamics of Forest Fragments Project (BDFFP).

pasture and regenerating forest, located 80 km north of Manaus, Amazonas, Brazil (Fig. 1) (60°00′W, 2°20′S) (Laurance et al., 2002; Lovejoy et al., 1986). These areas are protected by Brazilian law; they are legally under the responsibility of the Chico Mendes Institute for Biodiversity Protection (ICMBio) and are managed in partnership with the National Institute for Research in Amazonia (INPA).

The area is at 50–100 m elevation (Lovejoy et al., 1986). Rainforests in the area are *terra firme* (upland, not seasonally inundated). Species richness of trees is very high and can exceed 280 species ($\geqslant$10 cm DBH) per hectare (de Oliveira and Mori, 1999). Rainfall ranges from 1900 to 3500 mm annually with a pronounced dry season from June to October.

The soils in the study area are classified as yellow Oxisol according to RADAMBRASIL soilmaps (Fearnside and Leal Filho, 2001). These soils are heavily weathered, acidic, and very poor in nutrients such as P, Ca and K (Chauvel, 1982; Chauvel et al., 1987).

A soil survey in the BDFFP area included installation of a grid system of 1-ha permanent plots, each subdivided into 25 subplots measuring 20 × 20 m (Fearnside and Leal Filho, 2001). The vertices of each subplot were marked with PVC pickets tagged with an identifying letter and number. This grid creates a false coordinate system that allows the exact position of the data collection points to be located in the field. The aim was to systematize the monitoring of experiments and facilitate comparison of observed results.

2.2. Soil sampling

Soil samples were taken between 1984 and 1986 before isolation of the forest fragments, in the mid-1980s. These results were published by Fearnside and Leal Filho (2001). In the present study a new soil sample was collected between 2012 and 2013 at the location of each initial sample in order to allow comparison of the data series and to permit inferences about the effects of forest fragmentation on soil-carbon stocks over the long term.

The methodology of the second sampling followed the same collection protocol originally used except for the type of soil auger. Sampling covered a smaller number of plots. This study sampled seven isolated reserves, of which three were of 1 ha, two of 10 ha and two of 100 ha. Samples were collected from a total of 265 plots distributed over 28 ha (Supplementary Online Material: Table S1).

Soil samples were collected at 0–20 cm depth and were of two types: those for calculating carbon concentration and those for determining soil density. The samples for carbon were obtained using a screw auger; each individual sample was composed of five subsamples taken at each corner and in the center of the plot. A single soil-density sample was collected in the center of the plot using an auger especially designed to collect undisturbed samples. All samples were stored in plastic bags in the field until they were processed in the laboratory.

2.3. Sample treatment and analysis

2.3.1. Sample preparation

In the laboratory the samples for carbon quantification were dried in a solar oven for approximately one week, depending on the sample moisture content and on weather conditions. After drying, the samples were manually ground with a pestle and passed through sieves with 20-mm and 2-mm mesh, respectively. Plant roots and other visible fractions were then removed and set aside. A portion of each dried soil sample was stored in a glass flask in a collection of voucher specimens, allowing for comparisons among samples from different dates. A smaller portion of each sample was reduced to powder in an automatic grinder before being analyzed. Samples for calculating bulk density were placed in aluminum containers to dry in an electric oven at 105 °C for approximately 72 h until they reached constant weight.

2.3.2. Carbon

To determine the carbon content of samples collected in the 1980s, subsamples were taken from the voucher samples that had been stored in the soil collection. These were re-analyzed and compared with the current sample. This approach was used in order to eliminate or reduce any differences stemming from the analysis technique employed. Carbon contents (total C) were determined using a Vario MAX C/N elemental analyzer (Elementar Instruments, Hanau, Germany). In this apparatus the elements from samples are converted (by combustion) into gases that are mixed and kept at standard conditions of temperature and pressure; the gases are then depressurized in a column, and components are identified and separated based on thermal conductivity (Pérez et al., 2001). The samples were analyzed in the Laboratory of Soils and Plants at the National Institute for Research in Amazonia (INPA) in Manaus, Amazonas. The laboratory participates in inter-laboratory exchanges of samples and performs comparisons with standard samples; the laboratory has an "A" rating from the Brazilian Enterprise for Agricultural and Ranching Research (EMBRAPA).

2.3.3. Soil bulk density

Bulk density was determined by dividing the mass of the oven-dried soil sample by its volume (in the soil's natural state when the sample was taken with a volumetric soil sampler) and is expressed here in grams per cubic centimeter (g cm^{-3} = Mg m^{-3}). The bulk densities for the initial data series from this study were unavailable for some of the selected grids. Therefore, in order to avoid erroneous comparisons, only the density samples collected in 2012 were included in the analyses. These densities had a mean of 1.03 g cm^{-3} (SD = 0.21, n = 265). Our volumetric sampler had a volume of 392.5 cm^3.

2.3.4. Georeferencing

The grids of the permanent plots were georeferenced from Google Earth satellite images using the "Openlayer plugin" of Quantum GIS 1.8.0 (Lisbon, Portugal). Georeferencing was based on Datum WGS1984 and used the Universal Transverse Mercator (UTM) coordinate system.

2.4. Statistical analysis

2.4.1. Soil-carbon stock

The equation used to estimate soil-carbon stock was developed by Veldkamp (1994) as below:

$$CS = (TCC \times DS \times E) \times 10^4,$$

where

CS = Soil-carbon stock (Mg ha^{-1}),
TCC = Total carbon concentration (kg kg^{-1}),
BD = Bulk density (Mg m^{-3}),
D = Soil depth (m).

Soil-carbon stock changes were calculated as follows:

$$CSC = CS(t_1) - CS(t_0),$$

where

CSC = Soil-carbon stock changes (Mg ha^{-1}),
CS = Soil-carbon stock (Mg ha^{-1}),
t_0 = Beginning time,
t_1 = Ending time.

2.4.2. Geographically weighted regression (GWR)

Relations between changes in soil-carbon stocks and edge effects were quantified by means of a regression analysis plot (cf, Brunsdon et al., 1996, 1998; Fotheringham et al., 1996). The dependent variable was the soil-carbon content measured at each sampling point and the independent variable was the distance to the nearest edge of the fragment.

Geographically weighted regression (GWR) is a technique based on simple regression models; it differs from other techniques because it considers spatial variation in the set of relationships among variables. The basic difference between GWR and other regression models is that local regression coefficients (β) are generated for each sampled location (u). The regression coefficients do not remain fixed throughout the space, varying for each sampled location. Regression equations are generated for each location taking into account the spatial heterogeneity of the environments. In the GWR method, the relationship between proximity and similarity is considered, since different weights are assigned to different observations according to the degree of closeness (i.e., the closer the samples are to each other, the greater the weight assigned by the model). Samples from points closer to each other will have more in common than those collected far away from each other (Charlton and Fotheringham, 2009; Fotheringham et al., 2002). This tool has been shown to be a good estimator for geochemical parameters and for showing spatial patterns in response to changes in environmental factors (Zhang et al., 2011).

The GWR in this study was performed in ArcGIS software. In order to define the desired spatial relationship to explain our model, the kernel (the relative weight assignment system) used was of the bi-quadratic or "adaptative" type, i.e., the higher the density of sample points per unit area, the smaller the number of observations included in the estimate. The threshold distance used was 100 m, taking into account studies indicating that edge effects occur in this interval (Broadbent et al., 2008). The number of neighbors included around each observation was eight, considering that each sampling point can be represented as a pixel and that the pixel can be influenced by its eight direct neighbors.

Analysis and interpretation of the results of the GWR model considered the regression coefficient (r^2) together with the p value. Subsequently, homocedasticity of the regression residuals was evaluated by the Moran I spatial-autocorrelation test using ArcGIS software, and also by means of a graphical plot of the distribution of residuals versus the estimated values.

3. Results

Initial soil-carbon stocks ranged from 15.96 to 59.86 Mg ha^{-1}, with a mean of 33.41 ± 7.60 Mg ha^{-1}. Final stocks had wider variation, ranging from 7.44 to 60.97 Mg ha^{-1}, with a mean of 34.50 ± 8.34 Mg ha^{-1}. Comparison of soil-carbon stocks between the two sampled years indicated a significant difference (p = 0.01) in the 202 forest edge plots (⩽100 m from an edge), with a mean gain of 1.34 Mg ha^{-1} over the ~30-year period between the samplings (Table 1). There was no significant change in the 63 forest interior plots (Table 1). Note that in Table 1 the comparisons are between measurements before and after the ~30 year interval at the same location (i.e., comparisons between columns in the table), not comparisons between edge and interior plots (i.e., differences between rows).

Carbon content varied between 0.72 and 2.5%, with a mean of 1.67 ± 0.35% in the initial sample. In the most recent samples the range was from 0.72 to 3.06%, with a mean of 1.72%. Percent carbon content also increased significantly (p = 0.01) in the edge plots, but showed no significant change in the interior plots (Table 1).

The GWR results indicate a significant relationship between soil-carbon stock change, distance from the edge and clay content.

Table 1
Basic statistics for soil-carbon content and stock and their changes in forest remnants.[a]

	Soil-carbon content (%)			Soil-carbon stock (Mg ha^{-1})		
	Initial	Final	Gain	Initial	Final	Gain
Forest edge plots (⩽100 m from edge)						
Mean	1.66	1.72	0.06	33.09	34.43	1.34
Standard deviation	0.37	0.44	0.04	7.52	8.39	7.60
n	202	202	202	202	202	202
Paired *t*-test	p = 0.01			p = 0.01		
Forest interior plots (>100 m from edge)						
Mean	1.705	1.711	0.006	34.45	34.72	0.27
Standard deviation	0.31	0.30	0.03	7.81	8.27	6.07
n	63	63	63	63	63	63
Paired *t*-test	p = 0.86			p = 0.73		

[a] Note that the comparisons are between measurements before and after the ~30 year interval at the same location (i.e., comparisons between columns in the table), not comparisons between edge and interior plots (i.e., differences between rows).

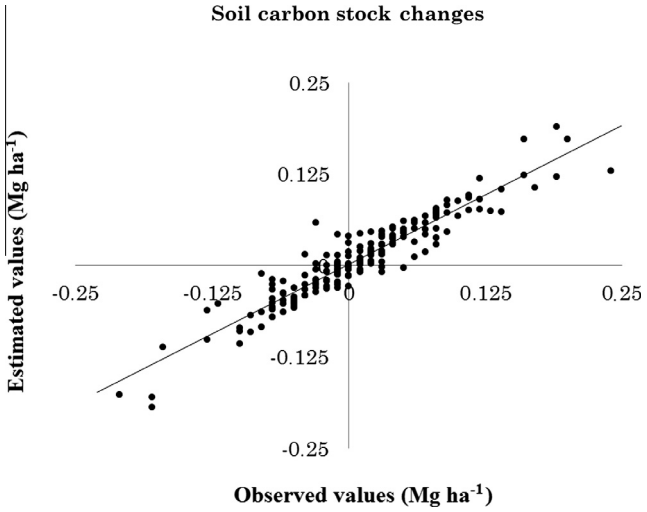

Fig. 2. Observed vs. estimated values (GWR).

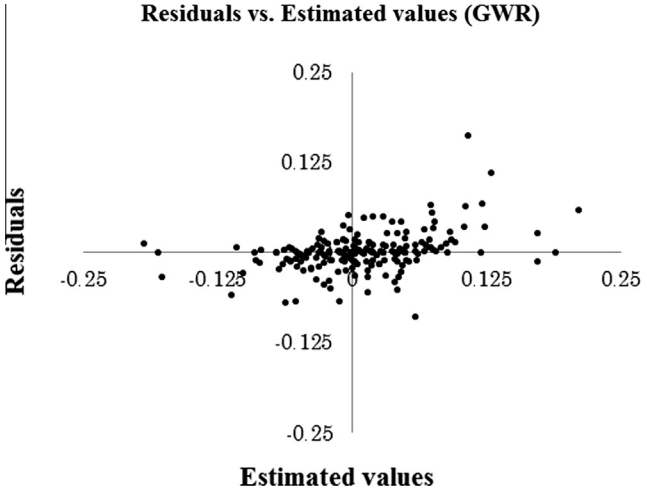

Fig. 3. Residuals vs. estimated values from geographically weighted regression (GWR) models.

The regression coefficient (r^2) indicates that 79% of the variation in the data was explained by this model (Fig. 2).

Plots of regression residuals versus fitted (predicted) values do not show homogeneous patterns in the distribution of points,

which allows us to affirm that the variance of the residuals is homocedastic and, therefore, the model has a satisfactory fit (Fig. 3). In addition, spatial-autocorrelation analysis (Moran's I) shows a random distribution of residuals with a Z value of −0.533, which is considered satisfactory (ESRI, 2010).

4. Discussion

The current study determined soil carbon using a Vario MAX C/N analyzer, which differs from the methods used in Fearnside and Leal Filho (2001). The earlier study reported both organic carbon (determined with the modified Walkley-Black method) and total carbon. The total carbon method was similar but not identical to the carbon determinations in the present study. In the earlier study, total carbon was determined at the National Center for Nuclear Energy in Agriculture (CENA) in Piracicaba, São Paulo, using the "dry" method, with combustion at 1100 °C followed by detection in a sodium chloride cell based on electrical conductivity (Cerri et al., 1990). The current study's reliance on simultaneous determinations using the same apparatus for both "before" and "after" samples avoids the possibility of differences in laboratory methods leading to spurious conclusions on carbon changes. Whether storage of the dry voucher samples in glass bottles for ~30 years could have resulted in any change in carbon content is unknown, but any such change would have been equal for samples from edge and interior plots.

The significant increase in soil-carbon stocks in forest fragments was contrary to our expectations. Originally we thought that the soil organic matter could be quickly decomposed as function of temperature increase in the fragment edges, leading to reduction of soil-carbon stocks in these areas. However the microclimatic changes reported after fragmentation, such as increased temperature, soil drying (Camargo and Kapos, 1995) and wind incidence (Laurance et al., 1998), are affecting soil-carbon stocks in a different way because these factors are directly related to increased mortality of adult trees at the edges of the fragments (Laurance et al., 1998). As a result of this mortality there is increased production and deposition of litter, as observed by other authors working at the same location (Laurance et al., 2006; Nascimento and Laurance, 2004, 2006; Sizer et al., 2000).

Nascimento and Laurance (2006) analyzed the effects of fragmentation on plant structure and found higher densities of pioneer species in fragments, averaging 48.9 individuals in plots located up to 300 m from the edge and 18.7 individuals in plots >300 m from the edge. In addition, larger amounts of dead biomass were observed in plots <300 m from the edge (43.9 Mg ha^{-1}) than in the forest interior plots (34.5 Mg ha^{-1}). The trees that died in forest edges during the first years after edge formation were of large diameter and high wood density, while those that replaced them were of smaller diameter and lower wood density (Nascimento and Laurance, 2004).

Chambers et al. (2000) calculated the rate of decomposition of dead trees and found that weight loss was faster in trunks that were thinner and less dense. Based on an average mortality of seven individuals ha^{-1} year^{-1}, production of coarse woody debris (>10 cm diameter) was estimated to be 3.6 Mg ha^{-1} year^{-1} with a residence time of approximately 5.9 years.

Thus, structural changes in the vegetation, such as the higher density of trees with thinner trunks and the larger proportion of pioneer species with lower density wood, decrease the carbon residence time in wood and accelerate decomposition, shifting the carbon flow from necromass to the soil compartment. The differences in soil-carbon stock change between the edges and the cores of the forest fragments must be occurring due to the increased litter decomposition rate at the edges, as suggested by Nascimento and Laurance (2004). These authors observed a negative relationship between the distance from the edge and the annual decomposition rate (turnover) of coarse woody debris (>10 cm diameter), with rates of 0.16 Mg ha^{-1} year^{-1} in the edge plots (up to 300 m from the forest edge) versus 0.12 Mg ha^{-1} year^{-1} in the fragment interiors (>300 m from the edge).

It is known that bulk density can be altered by a number of factors such as non-conservationist agricultural practices, mechanized silvicultural operations, forest-to-pasture conversion followed by cattle trampling, among other forms of disturbance (Cassel, 1983; Hamzaa and Anderson, 2005; Labelle and Jaeger, 2011; McGrath et al., 2001). It is also known that the areas where the BDFFP experiments were done have no history of use related to any of the activities mentioned above, with forest remaining standing except around the fragments where clearing was performed for conversion to cattle pasture. However, the addition of organic matter to a forest soil can affect bulk density over time because organic material is less dense than mineral soil (Asok and Sobha, 2014).

Didham (1998) found higher decomposition rates in fragments, as compared to continuous forests, thus giving support to the idea that carbon content in litter is flowing to the soil mineral fraction and resulting in carbon-stock increases in these areas. On the other hand, Vasconcelos and Laurance (2005) found that the decomposition rate of leaves in primary forest is greater than in secondary forests, and suggest that this is primarily because of the nutritional content of leaves in primary forest, with higher amounts of N and lower C:N ratios. This contradicts our idea that the higher rate of decomposition of necromass seen in the fragments could result in increased soil-carbon stocks. However, it is important to remember that, although the vegetation at forest edges may take on characteristics similar to those of secondary forests, the vegetation in fragments has not been exposed to clearcutting and this vegetation also has features in common with the primary forest.

Carbon-flow balance is positive in the soil in forest fragments, suggesting that carbon inputs derived from necromass decomposition (leaves, stems and roots) to the mineral soil outweigh the outputs from microbial respiration of carbon already stored in this compartment. Distance to the edge appears to have an indirect influence on this result due to microclimatic changes, leading to the increase in soil-carbon stocks at the edges of the fragments.

Considering 1 km as the width of edge effects, Skole and Tucker (1993) estimated that there were 341,000 km^2 of forest edges in Brazil's Legal Amazonia region in 1988. This is an area half the size of the US state of Texas. The portion of this 1-km width edge that is ⩽100 m from a clearing would be slightly less than 10% of the area, or 34,100 km^2. This edge area refers to edges where forest adjoins clearing for pasture and agriculture. Edges of this type are termed "hard edges" by Broadbent et al. (2008), who estimated that "soft edges" formed by forest near gaps created by selective logging each year represent an area over six times greater than the annual creation of "hard" edge areas. Broadbent et al. (2008) interpreted satellite imagery in four of the nine states of Brazil's Legal Amazonia region, but encompassing the "arc of deforestation" that accounts for 80% of the deforestation and logging activity. They estimated that in 2002 their study area had 368,150 km^2 of forest located less than 2 km from a forest edge (either hard or soft), or 53% of the 699,855 km^2 of forest that was still standing in their study area. The area within 1 km of an edge was 259,946 km^2 (37% of the forest), and within 100 m was 44,791 km^2 (6.4% of the forest). These figures make clear the importance of better quantification of changes in carbon balance and other features of edge areas.

A rough calculation of the scale of carbon impacts in Legal Amazonia can be done as follows. Considering only the change

we observed for ~30 years (Table 1), each km^2 of 100-m width edge gained 134 Mg of carbon in the top 20 cm of soil. Considering the 44,791 km^2 of 100-m width edges (either hard or soft) in 2002 in the portion of Brazilian Amazonia studied by Broadbent et al. (2008), this area would have gained 6.0×10^6 Mg of carbon in the top 20 cm of soil. Note, however, that many of the hard edges are cleared by advancing deforestation within a few years, and that carbon gain in this soil will be lost upon conversion to cattle pasture. The new equilibrium for soil carbon will be the lower one observed in pastures under typical (minimal) management (Fearnside and Barbosa, 1998).

"Biomass collapse" within 300 m of forest edges is estimated to have reduced aboveground live biomass by an average of 22.7 ± 31.98 Mg ha^{-1} in the BDFFP study area (Nascimento and Laurance, 2004), or 11.4 MgC ha^{-1} assuming 50% carbon content in biomass. About half of the aboveground live biomass loss up to 300 m occurs in the first 100 m: considering the relationship between distance to edge and change in aboveground live biomass in large ($\geqslant$10 cm DBH) trees in 50 1-ha plots (Nascimento and Laurance, 2004) and 50% carbon content in trees, 17.4 MgC ha^{-1} was lost in the first 100 m and 8.3 MgC ha^{-1} in the 100–300 m band in these plots 10–19 years after edge formation. Nascimento and Laurance (2004) found that biomass recovery from increased understory growth near edges was small (1.7 Mg ha^{-1} or 0.9 MgC ha^{-1} in the 0–300 distance range). Applying this value to the first 100 m, the net loss in live aboveground biomass is 16.6 MgC ha^{-1}. Nascimento and Laurance (2004) also found an increase in necromass (10.1 Mg ha^{-1} or 5.1 MgC ha^{-1} in the 0–300 distance range), but this carbon will be released as the necromass decays, with the exception of what is incorporated as soil carbon. Considering the 16.6-MgC ha^{-1} net loss of live aboveground biomass in the first 100 m, the 1.34 MgC ha^{-1} recovery we found in soil carbon to 20 cm depth would counterbalance 8.3% of this biomass collapse.

There is a need for more detailed information on the time path of carbon-stock changes in tropical forest areas, rather than simple difference calculations based on a long-term equilibrium condition. This has become evident since the 2009 Copenhagen agreement defined "dangerous" interference with the global climate system as exceeding 2 °C global mean temperature increase over the pre-industrial mean, and it is even more important considering the 1.5 °C aspiration of the 2015 Paris agreement. This implies the need for emissions estimates over short periods (ideally annual), rather than estimates of "net commited emissions" over the long term. Quantification of changes in both biomass and soil carbon stocks in forest edges is needed for these more-refined estimates.

While soil carbon increased in edges in our study, there was no significant change in the 63 forest-interior plots. This result is important in indicating lack of a readily detectible impact from global climate change on the Amazon forest's soil carbon compartment to date. Longterm monitoring of permanent plots such as these will be needed to track the impacts of future climate change.

5. Conclusions

Quantifying carbon-stock changes in forest edges, and similar changes under edge-like conditions in forests subjected to logging, represents a lacuna in many estimates of anthropogenic emissions from Amazon forest.

Carbon stocks increased in the soil up to 100 m from the edges of forest fragments after about 30 years of isolation.

Changes in soil-carbon stocks are related to the distance from the edge of the fragments, with greater changes observed in plots located closer to the edges.

This carbon storage in the soil compartment counteracts some (8.3%) of the impact of "biomass collapse" in Amazonian forest edges.

Soil carbon in the forest interior did not change significantly, suggesting that global climate change has not yet had a detectible effect on this compartment.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2016.08.002.

References

Asner, G.P., Broadbent, E.N., Oliveira, P.J.C., Keller, M., Knapp, D.E., Silva, J.N.M., 2006. Condition and fate of logged forests in the Brazilian Amazon. Proc. Nat. Acad. Sci. USA 103 (34), 12947–12950. http://dx.doi.org/10.1073/pnas.0604093103.

Asok, V.S., Sobha, V., 2014. Analysis of variation of soil bulk densities with respect to different vegetation classes, in a tropical rain forest – a study in Shendurney Wildlife Sanctuary, S. Kerala, India. Glob. J. Environ. Res. 8 (1), 17–20.

Batjes, N.H., 2005. Organic carbon stocks in the soils of Brazil. Soil Use Manag. 21, 22–24.

Batjes, N.H., Dijkshoorn, J.A., 1999. Carbon and nitrogen stocks in the soils of the Amazon region. Geoderma 89, 273–286.

Berenguer, E., Ferreira, J., Gardner, T.A., Aragão, L.E.O.C., de Camargo, P.B., Cerri, C.E., Durigan, M., de Oliveira Junior, R.C., Vieira, I.C.G., Barlow, J., 2014. A large-scale field assessment of carbon stocks in human-modified tropical forests. Glob. Change Biol. 20 (12), 3713–3726. http://dx.doi.org/10.1111/gcb12627.

Blanc, L., Echard, M., Herault, B., Bonal, D., Marcon, E., Chave, J., Baraloto, C., 2009. Dynamics of aboveground carbon stocks in a selectively logged tropical forest. Ecol. Applic. 19 (6), 1397–1404. http://dx.doi.org/10.1890/08-1572.1.

Brazil, INPE (Instituto Nacional de Pesquisas Espaciais), 2015. Projeto PRODES: Monitoramento da floresta amazônica brasileira por satélite. INPE, São José dos Campos, São Paulo, Brazil <http://www.obt.inpe.br/prodes/index.php> (accessed 14 December 2015).

Britto, A.S., 2015. Caracterização da Extração Seletiva de Madeira na Reserva de Desenvolvimento Sustentável do Juma e Sua Zona de Amortecimento. Masters thesis in Management of Amazonian Protected Areas, Instituto Nacional de Pesquisas da Amazônia (INPA), Manaus, Amazonas, Brazil. 87 pp.

Broadbent, E.N., Asner, G.P., Keller, M., Knapp, D.E., Oliveira, P.J.C., Silva, J.N., 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. Biol. Conserv. 141, 1745–1757. http://dx.doi.org/10.1016/j.biocon.2008.04.024.

Brunsdon, C., Fotheringham, A.S., Charlton, M., 1996. Geographically weighted regression: a method for exploring spatial non-stationarity. Geograp. Anal. 28 (4), 281–298.

Brunsdon, C., Fotheringham, A.S., Charlton, M., 1998. Geographically weighted regression-modelling spatial non-stationarity. Statistician 47 (3), 431–443.

Camargo, J.L.C., Kapos, V., 1995. Complex edge effects on soil moisture and microclimate in central Amazonian forest. J. Trop. Ecol. 11, 205–221.

Cassel, D.K., 1983. Spatial and temporal variability of soil physical-properties following tillage of Norfolk loamy sand. Soil Sci. Soc. Am. J. 7, 96–201.

Cerri, C.C., Eduardo, B.P., Piccolo, M.C., 1990. Métodos de Análises em Matéria Orgânica do Solo. Centro de Energia Nuclear na Agricultura/Universidade de São Paulo (CENA/USP), Piracicaba, São Paulo, Brazil, 78 pp.

Chambers, J.Q., Higuchi, N., Schimel, J.P., Ferreira, L.V., Melack, J.M., 2000. Decomposition and carbon cycling of dead trees in tropical forests of the central Amazon. Oecologia 122, 380–388.
Charlton, M., Fotheringham, A.S., 2009. Geographically Weighted Regression White Paper. National Centre for Geocomputation, National University of Ireland, Maynooth, Ireland, 17 pp. <http://www.geos.ed.ac.uk/~gisteac/fspat/gwr/gwr_arcgis/GWR_WhitePaper.pdf> (accessed 14 December 2015).
Chauvel, A., 1982. Os latossolos amarelos, alicos, argilosos dentro dos ecossistemas das bacias experimentais do INPA e da regiao vizinha. Acta Amazon. 12 (3), 38–47.
Chauvel, A., Lucas, Y., Boule, R., 1987. On the genesis of the soil mantle of the region of Manaus, Central Amazonia, Brazil. Experientia 43, 234–241.
de Oliveira, L.A.A., Mori, S.A., 1999. A central Amazonian terra firme forest. I. High tree species richness on poor soils. Biodivers. Conserv. 8, 1219–1244.
Didham, R.K., 1998. Altered leaf-litter decomposition rates in tropical forest fragments. Oecologia 116, 397–406.
ESRI (Environmental Systems Research Institute, Inc.), 2013. ArcGIS Desktop Help 10.1 (accessed 15 December 2015).
Fearnside, P.M., 1997. Greenhouse gases from deforestation in Brazilian Amazonia: net committed emissions. Clim. Change 35 (3), 321–360. http://dx.doi.org/10.1023/A:1005336724350.
Fearnside, P.M., 2000a. Greenhouse gas emissions from land-use change in Brazil's Amazon region. In: Lal, R., Kimble, J.M., Stewart, B.A. (Eds.), Global Climate Change and Tropical Ecosystems. Advances in Soil Science. CRC Press, Boca Raton, Florida, pp. 231–249.
Fearnside, P.M., 2000b. Global warming and tropical land-use change: greenhouse gas emissions from biomass burning, decomposition and soils in forest conversion, shifting cultivation and secondary vegetation. Clim. Change 46 (1–2), 115–158. http://dx.doi.org/10.1023/A:1005569915357.
Fearnside, P.M., 2015. Deforestation soars in the Amazon. Nature 521, 423. http://dx.doi.org/10.1038/521423b.
Fearnside, P.M., 2016. Brazil's Amazonian forest carbon: The key to Southern Amazonia's significance for global climate. Reg. Environ. Change. http://dx.doi.org/10.1007/s10113-016-1007-2.
Fearnside, P.M., Barbosa, R.I., 1998. Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia. For. Ecol. Manage. 108 (1–2), 147–166. http://dx.doi.org/10.1016/S0378-1127(98)00222-9.
Fearnside, P.M., Leal Filho, N., 2001. Soil and development in Amazonia: lessons from the biological dynamics of forest fragments project. In: Bierregaard, R.O., Gascon, C., Lovejoy, T.E., Mesquita, R. (Eds.), Lessons from Amazonia: The Ecology and Conservation of a Fragmented Forest. Yale University Press, New Haven, Connecticut, pp. 291–312.
Fearnside, P.M., Righi, C.A., Graça, P.M.L.A., Keizer, E.W.H., Cerri, C.C., Nogueira, E.M., Barbosa, R.I., 2009. Biomass and greenhouse-gas emissions from land-use change in Brazil's Amazonian "arc of deforestation": the states of Mato Grosso and Rondônia. For. Ecol. Manage. 258, 1968–1978. http://dx.doi.org/10.1016/j.foreco.2009.07.042.
Fonseca, A., Souza Jr., C., Veríssimo, A., 2015. Boletim do Desmatamento da Amazônia Legal (Outubro De 2015) SAD. Instituto do Homem e Meio Ambiente da Amazônia (IMAZON), Belém, Pará, Brazil, 10 pp <http://imazon.org.br/PDFimazon/Portugues/transparencia_florestal/amazonia_legal/SAD%20outubro%202015.pdf> (accessed 18 December 2015).
Fotheringham, A.S., Brunsdon, C., Charlton, M.E., 2002. Geographically Weighted Regression: The Analysis of Spatially Varying Relationships. Wiley, Chichester, UK.
Fotheringham, A.S., Charlton, M.E., Brunsdon, C., 1996. The geography of parameter space: an investigation of spatial non-stationarity. Int. J. Geogr. Inform. Syst. 10 (5), 605–621.
Gerwing, J.J., 2002. Degradation of forests through logging and fire in the eastern Brazilian Amazon. For. Ecol. Manage. 157 (1), 131–141. http://dx.doi.org/10.1016/s0378-1127(00)00644-7.
Hamzaa, M.A., Anderson, W.K., 2005. Soil compaction in cropping systems: a review of the nature, causes and possible solutions. Soil Till. Res. 82 (2), 121–145.
Jackson, S.M., Fredericksen, T.S., Malcolm, J.R., 2002. Area disturbed and residual stand damage following logging in a Bolivian tropical forest. For. Ecol. Manage. 166, 271–283. http://dx.doi.org/10.1016/S0378-1127(01)00681-8.
Labelle, E.R., Jaeger, D., 2011. Soil compaction caused by cut-to-length forest operations and possible short-term natural rehabilitation of soil density. Soil Sci. Soc. Am. J. 75 (6), 2314–2329.
Laurance, W.F., Andrade, A.S., Magrach, A., Camargo, J.L.C., Valsko, J.J., Campbell, M., Fearnside, P.M., Edwards, W., Lovejoy, T.E., Laurance, S.G., 2014a. Long-term changes in liana abundance and forest dynamics in undisturbed Amazonian forests. Ecology 95 (6), 1604–1611. http://dx.doi.org/10.1890/13-1571.1.
Laurance, W.F., Andrade, A.S., Magrach, A., Camargo, J.L.C., Valsko, J.J., Campbell, M., Fearnside, P.M., Edwards, W., Lovejoy, T.E., Laurance, S.G., 2014b. Apparent environmental synergism drives the dynamics of Amazonian forest fragments. Ecology 95 (11), 3018–3026. http://dx.doi.org/10.1890/14-0330.1.
Laurance, W.F., Bierregaard, R.O. (Eds.), 1997. Tropical Forest Remnants: Ecology Management and Conservation of Fragmented Communities. University of Chicago Press, Chicago, Illinois, U.S.A..
Laurance, W.F., Camargo, J., Luizão, R.C.C., Laurance, S.G.W., Pimm, S.L., Bruna, E., Stouffer, P.C., Williamson, G.B., Benitez-Malvido, J., Vasconcelos, H., Van Houtan, K., Zartman, C.E., Boyle, S., Didham, R.K., Andrade, A., Lovejoy, T.E., 2011. The fate of Amazonian forest fragments: a 32-year investigation. Biol. Conserv. 144, 56–67.
Laurance, W.F., Ferreira, L.V., Rankin-de-Merona, J.M., Laurance, S.G., 1998. Rain forest fragmentation and the dynamics of Amazonian tree communities. Ecology 69, 2032–2040.
Laurance, W.F., Laurance, S.G., Ferreira, L.V., Rankin-de Merona, J., Gascon, C., Lovejoy, T.E., 1997. Biomass collapse in Amazonian forest fragments. Science 278, 1117–1118.
Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bierregaard, R.O., Laurance, S.G.W., Sampaio, E., 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. Conserv. Biol. 16, 605–618.
Laurance, W.F., Nascimento, H.E.M., Laurance, S.G., Andrade, A., Ribeiro, J.E.L.S., Giraldo, J.P., Lovejoy, T.E., Condit, R., Chave, J., D'Angelo, S., 2006. Rapid decay of tree-community composition in Amazonian forest fragments. Proc. Nat. Acad. Sci. USA 103 (50), 19010–19014.
Lovejoy, T.E., Bierregaard, R.O.J., Rylands, A.B., Malcolm, J.R., Quintela, C.E., Harper, L.H., Brown, K.S.J., Powell, A.H., Powell, G.V.N., Schubart, H.O.R., Hays, M.B., 1986. Edge and other effects of isolation on Amazon forest fragments. In: Soule, M.E. (Ed.), Conservation Biology: The Science of Scarcity – Amazon Forest Dynamics and Diversity. Sinauer Associates, Sunderland, Massachusetts, pp. 257–285.
McGrath, D.A., Smith, C.K., Gholz, H.L., Oliveira, F.A., 2001. Effects of land-use change on soil nutrient dynamics in Amazônia. Ecosystems 8 (4), 625–645.
Monteiro, A., Cardoso, D., Conrado, D., Veríssimo, A., Souza Jr., C., 2013. Transparência Manejo Florestal Estado do Pará – 2011 a 2012. Instituto do Homem e Meio Ambiente da Amazônia (IMAZON), Belém, Pará, Brazil. 14 pp. <http://imazon.org.br/PDFimazon/Portugues/transparencia_manejo_florestal/BTMF_Para_21out2013.pdf> (accessed 15 December 2015).
Murcia, C., 1995. Edge effects in fragmented forests: implications for conservation. Trends Ecol. Evol. 10, 58–62.
Nascimento, H.E.M., Laurance, W.F., 2004. Biomass dynamics in Amazonian forest fragments. Ecol. Appl. 14 (3), S127–S138.
Nascimento, H.E.M., Laurance, W.F., 2006. Efeitos de área e de borda sobre a estrutura em fragmentos de floresta de terra-firme após 13–17 anos de isolamento. Acta Amazon. 36 (2), 183–192.
Nogueira, E.M., Fearnside, P.M., Nelson, B.W., Barbosa, R.I., Keizer, E.W.H., 2008. Estimates of forest biomass in the Brazilian Amazon: new allometric equations and adjustments to biomass from wood-volume inventories. For. Ecol. Manage. 256 (11), 1853–1857. http://dx.doi.org/10.1111/gcb.12798.
Nogueira, E.M., Yanai, A.M., Fonseca, F.O.R., Fearnside, P.M., 2015. Carbon stock loss from deforestation through 2013 in Brazilian Amazonia. Glob. Change Biol. 21, 1271–1292. http://dx.doi.org/10.1111/gcb.12798.
Palace, M., Keller, M., Asner, G.P., Silva, J.N.M., Passos, C., 2007. Necromass in undisturbed and logged forests in the Brazilian Amazon. For. Ecol. Manage. 238, 309–318. http://dx.doi.org/10.1016/j.foreco.2006.10.026.
Pérez, D.V., de Alcântara, S., Arruda, R.J., Meneghelli, N.A.C., 2001. Comparing two methods for soil carbon and nitrogen determination using selected Brazilian soils. Commun. Soil Sci. Plant Anal. 32 (1 & 2), 295–309.
Rankin-de-Merona, J.M., Hutchings, R.W., 2001. Deforestation effects at the edge of an Amazonian Forest fragment: tree mortality, damage and recruitment. In: Bierregaard, R.O., Jr., Gascon, C., Lovejoy, T.E., Mesquita, R.C. (Eds.), Lessons From Amazonia: The Ecology and Conservation of a Fragmented Forest. Yale University Press, New Haven, Connecticut, pp. 107–120.
Saatchi, S., Harris, S.N.L., Brown, S., Lefsky, M., Mitchard, E.T.A., Salas, W., Zutta, B.R., Buermann, W., Lewis, S.L., Hagen, S., Petrova, S., White, L., Silman, Morel, M.A., 2011. Benchmark map of forest carbon stocks in tropical regions across three continents. Proc. Nat. Acad. Sci. USA 108 (24), 9899–9904.
Saunders, D.A., Hobbs, R.J., Margules, C.R., 1991. Biological consequences of ecosystem fragmentation: a review. Conserv. Biol. 5 (1), 18–32.
Silgueiro, V., Thuault, A., Micol, L., Abad, R., 2015. Mapeamento da ilegalidade na exploração madeireira entre agosto de 2012 e julho de 2013. Transparência Florestal Mato Grosso 4 (5), 1–12, <http://www.icv.org.br/site/wp-content/uploads/2015/05/transparencia_florestal_mapeamento_ilegalidade_madeireira.pdf> (accessed 18 December 2015).
Sist, P., Ferreira, F.N., 2007. Sustainability of reduced-impact logging in the Eastern Amazon. For. Ecol. Manage. 243, 199–209. http://dx.doi.org/10.1016/j.foreco.2007.02.014.
Sizer, N.C., Tanner, E.V.J., Ferraz, I.D.K., 2000. Edge effects on litterfall mass and nutrient concentrations in forest fragments in central Amazonia. J. Trop. Ecol. 16, 853–863.
Skole, D., Tucker, C., 1993. Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978 to 1988. Science 260, 1905–1909.
Vasconcelos, H.L., Laurance, W.F., 2005. Influence of habitat, litter type and soil invetebrates on leaf-litter decomposition in a fragmented Amazonian landscape. Oecologia 143, 456–462.
Vasconcelos, H.L., Luizão, F.J., 2004. Litter production and litter nutrient concentrations in a fragmented Amazonian landscape. Ecol. Appl. 14, 884–892.
Veldkamp, E., 1994. Organic carbon turnover in three tropical soils under pasture after deforestation. Soil Sci. Soc. Am. J. 58, 175–180.
Zhang, C., Tang, Y., Xu, X., Kiely, G., 2011. Towards spatial geochemical modelling: use of geographically weighted regression for mapping soil organic carbon contents in Ireland. Appl. Geochem. 26, 1239–1248.