THE ENVIRONMENTAL IMPACT OF SELECTIVE LOGGING

OPERATIONS IN RAINFOREST OF SOUTHERN CAMEROON: SOIL

DISTURBANCE, NUTRIENT BUDGET AND GREENHOUSE GAS FLUXES

Dissertation

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To my husband, Isaac Blaise, and my children: Manoel, Asaph and Hermann DJOKO

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SUMMARY

Tropical forests play an important role in climate change mitigation through carbon sequestration in the vegetation and soils, which can be released to the atmosphere through deforestation and forest degradation. One of the key drivers of tropical forest degradation is selective logging, which is the most common timber harvesting practice in the tropics. Selective logging in Cameroon involves harvesting only trees of commercially important species that have reached the minimum harvestable diameter at breast height, as defined by the forest administration. In general, less than one tree is harvested per hectare due to high forest diversity and limited markets for most timber species. Selective-logging operations result in four types of disturbed strata (felling gaps, skidding trails, logging decks and logging roads) that differ in the degree of disturbance. Besides these disturbed strata are patches of intact forest whose spatial extent depends on the harvest intensity. Yet, little is known about the impacts of selective logging on highly weathered soils of Congo Basin rainforests.

This thesis sets up to assess the impacts of selective logging on forest soils by estimating (1) the area directly affected by logging operations and nutrient export with timber harvest, (2) spatially explicit changes in soil physical and biochemical characteristics and (3) changes in soil greenhouse gas (GHG) fluxes. The study was conducted in two forests logged with conventional selective logging (CL) and reduced-impact logging (RIL) protocols, respectively, in southern Cameroon. In this study, CL refers to unplanned logging operations in an unmanaged forest while RIL implies well-planned operations carried out by trained crews in a logging concession certified for sustainable forest management. Both forests (CL and RIL) had comparable vegetation structure and soil characteristics prior to logging, and similar logging intensity (i.e., removals of < 0.3 tree ha^{-1}, equivalent to < 3 m^3 wood ha^{-1}). We designed our study such that each of the four replicate plots at each logging system covered

the four disturbed strata (road, logging deck, skidding trail and felling gap) and an adjacent undisturbed area as reference.

For our first objective, we conducted ground mapping to estimate the areal coverage of each disturbed stratum in both logging systems, using handheld global positioning system (GPS) and tape measure. Maps of the location and spatial extent of all felling gaps, skidding trails, logging decks and roads within CL and RIL were produced in QGIS. The ground area directly affected by logging operations accounted for less than 6% of the total forest area in both CL and RIL. We attributed the comparable ground disturbance between the two logging systems to the low logging intensity and the recruitment of local workers with prior RIL experience by CL operator. Due to low logging intensity, the amount of nutrients exported with harvested timber was lower compared to logging-induced soil nutrient losses. The total biomass-C emission from both selective logging systems (i.e., C in extracted timber and logging damage to residual stand) was estimated to be < 8 Mg biomass-C ha^{-1}, which represented $< 5\%$ of the pre-logging aboveground carbon stock.

To achieve our second objective, we determined post-logging soil physical and biochemical characteristics down to 50-cm depth at CL and RIL. We observed different changes in soil characteristics among the disturbed strata compared to the reference, with roads and logging decks being the most affected strata. Area-weighted average of the disturbed strata at CL and RIL showed overall reductions of 21–29% in SOC, N and P stocks relative to the reference areas. We attributed the reduction in element stocks to the removal of organic matter particularly in creating roads and logging decks during logging operations and the absence of plant material inputs after logging. At the scale of the logged forest, SOC, N and P losses equaled 0.9–1.5% of their stocks in the undisturbed area, and may be recovered within 14 years after logging.

Finally, we investigated changes in soil GHG fluxes following CL and RIL, and determined their controlling factors. Soil GHG fluxes were measured monthly from September 2016 to October 2017, using static vented chambers. Changes in soil GHG fluxes followed the spatial pattern of disturbance from selective logging with highest changes in roads, logging decks and skidding trails. Soil CO_2 emissions decreased in the disturbed area, following the alteration of SOC and nutrient availability, and the increase in water-filled pore space (WFPS). However, this does not reflect a reduction in net ecosystem CO_2 emissions following logging disturbance. We found increased soil N_2O emissions mainly from the skidding trails, and for soil CH_4, emissions from the disturbed area were seven times higher than consumption in the undisturbed reference area. Changes in soil N_2O and CH_4 fluxes were mainly controlled by soil N availability and WFPS. At the scale of the logged forest, soil CO_2 emissions and CH_4 uptake decreased by 1.3 and 36%, while soil N_2O emissions increased by 3.3% relative to undisturbed forest at CL and RIL.

The research presented in this dissertation shows that low-intensity selective logging can be a sustainable way to manage tropical forest if associated with a sufficient rotation time. As discussed in the synthesis chapter, increase in logging intensity and frequent logging may expand ground disturbance and consequently intensify changes in soil characteristics and GHG fluxes in logged forests. This suggests that though low-intensity CL is initially comparable to RIL, repeated logging operations may expose CL forest to greater degradation.

ZUSAMMENFASSUNG

Tropische Wälder spielen eine wichtige Rolle im Klimaschutz aufgrund der Speicherung von Kohlenstoff in der Vegetation und im Boden, welcher durch Abholzung und Waldschädigung an die Atmosphäre freigesetzt werden kann. Einer der Haupttreiber der Zerstörung tropischer Wälder ist der selektive Holzeinschlag, der in den Tropen die häufigste Holzerntepraxis darstellt. Beim selektiven Holzeinschlag in Kamerun werden nur Bäume von wirtschaftlich wichtigen Arten geerntet, die den von der Forstverwaltung festgelegten Mindestdurchmesser in Brusthöhe erreicht haben. Im Allgemeinen wird weniger als ein Baum pro Hektar geerntet, was auf eine hohe Walddiversität und begrenzte Märkte für die meisten Holzarten zurückzuführen ist. Selektive Holzeinschlagsaktivitäten führen zu vier Typen von gestörten Bereichen (Hiebslücken, Rückegassen, Holzlagerplätze und Holzabfuhrstraßen), die sich im Grad der Störung unterscheiden. Daneben gibt es intakte Waldflächen, deren räumliche Ausweitung von der Ernteintensität abhängt. Über die Auswirkungen des selektiven Holzeinschlags auf die hochverwitterten Böden der Regenwälder des Kongobeckens ist jedoch wenig bekannt.

Diese Arbeit wurde erstellt, um die Auswirkungen des selektiven Holzeinschlags auf Waldböden zu bewerten durch (1) die Beurteilung der Fläche die direkt von Holzeinschlagsaktivitäten und Nährstoffexport durch Holzernte betroffen ist, (2) Untersuchung von räumlich expliziten Veränderungen der bodenphysikalischen und biochemischen Eigenschaften und (3) Untersuchung der Veränderungen von Treibhausgasflüssen im Boden. Diese Studie wurde im Süden Kameruns in zwei Wäldern durchgeführt, deren Holzeinschlag mit konventionell selektivem Holzeinschlag (*conventional selective logging* (CL)) bzw. reduziertem Holzeinschlag (*reduced-impact logging* (RIL)) durchgeführt wurde. In dieser Studie bezieht sich CL auf ungeplante Holzeinschläge in einem

nicht bewirtschafteten Wald, wohingegen RIL sorgfältig geplante Holzeinschläge von ausgebildetem Personal in einer für nachhaltige Forstwirtschaft zertifizierten Forstkonzession impliziert. Beide Wälder (CL und RIL) wiesen vor dem Holzeinschlag eine vergleichbare Vegetationsstruktur und Bodeneigenschaften sowie eine ähnliche Abholzungsintensität (d.h. die Entfernung von unter 0,3 Bäumen pro Hektar, entsprechend weniger als 3 m^3 Holz pro Hektar) auf. Wir haben unsere Studie so konzipiert, dass jede der vier replizierten Parzellen in jedem Holzerntesystem die vier gestörten Bereiche (Abfuhrstraße, Holzlagerplatz, Rückegasse und Hiebslücke) abdeckte und ein angrenzender ungestörter Bereich als Referenz herangezogen wurde.

Für unsere erste Zielsetzung haben wir Bodenkartierungen durchgeführt um die flächenmäßige Abdeckung jedes gestörten Bereiches in beiden Holzerntesystemen mittels eines tragbaren globalen Positionierungssystems (GPS) und eines Maßbandes zu schätzen. In QGIS wurden Karten der Lage und räumlichen Ausdehnung aller Hiebslücken, Rückegassen, Holzlagerplätze und Abfuhrstraßen innerhalb von CL und RIL erstellt. Die direkt von den Holzeinschlägen betroffene Bodenfläche machte weniger als 6% der gesamten Waldfläche in CL und RIL aus. Die vergleichbare Bodenstörung zwischen den beiden Systemen führten wir auf die geringe Holzernteintensität und die Rekrutierung von lokalen Arbeitern mit vorheriger RIL-Erfahrung durch den CL-Betreiber zurück. Aufgrund der geringen Holzernteintensität war die Menge der mit geerntetem Holz exportierten Nährstoffe geringer als die durch die Holzernte verursachten Nährstoffverluste im Boden. Die gesamte Biomasse-Kohlenstoff-Emission aus beiden selektiven Holzerntesystemen (d.h. Kohlenstoff in geernteten Holz und Holzschäden am Restbestand) wurde auf unter 8 Mg Biomasse-C pro Hektar geschätzt, was weniger als 5% des oberirdischen Kohlenstoffvorrats vor dem Holzeinschlag entspricht.

Für unsere zweite Zielsetzung haben wir die physikalischen und biochemischen Bodeneigenschaften nach Holzeinschlag bis zu einer Tiefe von 50 cm für CL und RIL

bestimmt. Unterschiedliche Veränderungen in den Bodeneigenschaften der gestörten Bereiche konnten im Vergleich zur Referenz beobachtet werden, wobei die Abfuhrstraßen und Holzlagerplätze die am stärksten betroffenen Bereiche waren. Der flächengewichtete Durchschnitt der gestörten Bereiche bei CL und RIL zeigte insgesamt einen Rückgang der organischen Kohlenstoff (SOC)-, N- und P-Vorräte im Boden um 21–29% im Vergleich zu den Referenzgebieten. Wir führten diesen Rückgang auf die Entfernung von organischer Substanz zurück, insbesondere bei der Erstellung von Abfuhrstraßen und Holzlagerplätzen während der Holzernte, sowie auf das Fehlen der Zugabe von Pflanzenmaterial nach der Holzernte. In der Größenordnung des geschlagenen Waldes betrugen die Verluste von SOC, N und P 0,9–1,5% ihrer Vorräte im ungestörten Gebiet, sie können innerhalb von 14 Jahren nach dem Holzeinschlag wieder aufgefüllt werden.

Schließlich untersuchten wir Veränderungen der Treibhausgasflüsse im Boden nach CL und RIL und ermittelten deren Einflussfaktoren. Die Treibhausgasflüsse im Boden wurden monatlich von September 2016 bis Oktober 2017 mit statischen, belüfteten Hauben gemessen. Veränderungen der Treibhausgasflüsse im Boden folgten dem räumlichen Störungsmuster des selektiven Holzeinschlages mit den größten Änderungen in Abfuhrstraßen, Holzlagerplätzen und Rückegassen. Die CO_2-Emissionen im Boden verringerten sich im gestörten Gebiet, folgend den Änderungen in der SOC- und Nährstoffverfügbarkeit sowie der Vergrößerung des wassergefüllten Porenraumes. Dies spiegelt jedoch nicht eine Verringerung der Netto-Ökosystememissionen von CO_2 nach Störung durch Holzeinschlag wider. Wir stellten erhöhte N_2O-Emissionen des Bodens, hauptsächlich ausgehend von den Rückegassen fest. Die CH_4-Bodenemissionen im gestörten Gebiet waren siebenmal höher als der Verbrauch im ungestörten Referenzbereich. Änderungen der N_2O- und CH_4-Flüsse im Boden wurden hauptsächlich durch die Verfügbarkeit von Bodenstickstoff und dem wassergefülltem Porenraum gesteuert. Auf Ebene des geernteten Waldes sanken die CO_2-Emissionen des

Bodens und die Aufnahme von CH_4 um 1,3 bzw. 36%, während die N_2O-Emissionen um 3,3%

anstiegen, im Vergleich zu ungestörtem Wald bei CL und RIL.

Die in dieser Dissertation vorgestellten Ergednisse zeigen, dass der selektive Holzeinschlag geringer Intensität ein nachhaltiger Weg zur Bewirtschaftung von Tropenwald sein kann, wenn dieser mit einer ausreichenden Umtriebszeit verbunden ist. Wie im Synthesekapitel erläutert, können eine Erhöhung der Holzernteintensität und häufiger Holzeinschlag die Bodenstörung verstärken und somit die Veränderungen der Bodeneigenschaften und Treibhausgasflüsse in abgeholzten Wäldern verstärken. Dies deutet darauf hin, dass, obwohl CL mit niedriger Intensität zunächst mit RIL vergleichbar ist, wiederholte Holzeinschläge den CL-Wald einer größeren Degradation aussetzen können.

CHAPTER 1

GENERAL INTRODUCTION

1.1.   Selective logging in the tropics and its environmental impacts

Tropical forests provide a wide range of ecosystem services including biodiversity conservation and climate change mitigation through carbon sequestration. Over the past decades, the area of natural tropical forest has decreased due to human activities (Keenan et al., 2015), with the remaining forest currently in various degree of degradation (Putz and Romero, 2015). Among the main causes of tropical forest degradation are timber and fuelwood harvesting, understory wildfire and livestock grazing in the forest (Hosonuma et al., 2012). Globally, more than 400 million ha, which constitutes 22% of remaining world tropical forests, are allocated for timber production (Blaser et al., 2011; Keenan et al., 2015). Because only certain species of these species-diverse tropical forests are of commercial interest, selective logging has been the main timber harvesting method (Asner et al., 2005). Selective logging is the process of felling and extracting only few large trees of valuable species from natural production forests. This method of forest harvesting leaves behind four types of disturbed strata (i.e., felling gap, skidding trail, logging deck and road) and large patches of the logging block undisturbed (i.e., with no direct impact of logging on the vegetation and soils). The extent of intact forest within logging blocks are related to the logging intensity and even where this is high, intact forest patches can still be found in areas of steep slope and riparian buffers (Putz et al., 2019). Selective logging is widely considered as the better option in protecting the forest compared to clearcutting (Putz et al., 2008a). Indeed, one time selective logging of a forest can retain 47 to 97% of the pre-logging aboveground carbon depending on harvest intensity and practices (Putz et al., 2012). However, the effectiveness of selective logging in sustainable forest management remains an issue (Hari Poudyal et al., 2018), as unsustainable selective logging is the primary cause of tropical forest degradation (Pearson et al., 2017).

Many studies have been carried out on selective logging and its impacts across the tropics. Most of them (> 38%) focused on plant and animal species composition, diversity and richness, mainly in Brazil, Malaysia, Indonesia and Australia (Hari Poudyal et al., 2018). Other documented impacts include depletion of aboveground carbon stock and modification of wildlife habitat which contribute to tropical forest degradation (Martin et al., 2015; Pearson et al., 2014). Case studies have often reported conflicting findings on the impacts of selective logging on biodiversity. This reflects the variation in logging intensity (i.e., number of trees extracted per ha), as the magnitude of selective logging impact generally increases with logging intensity (Burivalova et al., 2014; Putz et al., 2012). Selective logging can also alter the forest structure and floristic composition (e.g., Asase et al., 2012; Clark and Covey, 2012). Direct impacts include decreases in stem density, total basal area and aboveground biomass (e.g., Blanc et al., 2009; Bonnell et al., 2011; Hall et al., 2003; Osazuwa-Peters et al., 2015). Thus, the consequent decrease in forest timber stock may not rebound to primary-forest level. A meta-analytical study on > 100 articles showed that timber yields can be reduced by about 46% after the first logging cycle (Putz et al., 2012). Indeed, canopy-disturbance associated with low-intensity selective logging is generally not enough to allow the regeneration of some valuable timber species, especially light-demanding trees (Fredericksen and Putz, 2003; Karsenty and Gourlet-Fleury, 2006). This may affect the economical sustainability of the forest if new markets do not open up for a greater number of species.

Less than 4% of studies on selective logging the impacts focused on forest soils (Hari Poudyal et al., 2018). Results of these studies show that selective logging can alter soil organic carbon and nutrient stocks (Bol and Tokuchi, 2018; Chiti et al., 2016; Durigan et al., 2017; McNabb et al., 1997; Olander et al., 2005; Vaglio Laurin et al., 2016), soil microbial biomass and composition (Mori et al., 2017; Tripathi et al., 2016), and soil greenhouse gas (GHG) fluxes (Keller et al., 2005; Mori et al., 2017; Yashiro et al., 2008).

1.2.   Conventional versus reduced-impact selective logging

Conventional logging is the oldest and most common selective logging practices in the tropics (Hari Poudyal et al., 2018). It refers to unplanned logging operations by untrained crews, which is assimilated to a "hit-or-miss timber hunting" (Holmes et al., 2002). Since there is no pre-logging forest inventory, conventional loggers locate merchantable trees and fell them with little regard to the residual stand. Moreover, without a detailed harvest plan, skidding crews travel within the forest to locate the logs, which results in significant damage to the residual stand and forest soils (Holmes et al., 2002). Conventional logging (CL) can therefore result in substantial forest degradation (Medjibe and Putz, 2012). Reduced-impact logging (RIL) was introduced in attempt to reduce selective logging-associated damage to soil, residual stands, and workers, and to improve efficiency in timber harvesting (Putz et al., 2012). In contrast to CL, RIL consists of careful planning and control of timber harvesting operations, which are conducted by trained and supervised crews (Pinard and Putz, 1996; Putz et al., 2008a). In general, RIL techniques are advocated as a key tool towards sustainable forest management (Putz et al., 2012). They include pre-logging forest inventory, stand mapping, skidding trail planning, future crop tree flagging, improved felling techniques and post-logging assessments (FAO, 2005; Putz et al., 2008a).

Compared to conventional selective logging, RIL can generate significant ecological benefits. For example, RIL has been effective in reducing ground area damage (Pereira Jr. et al., 2002; van der Hout, 2000), incidental damage to the remaining stand (Martin et al., 2015; Medjibe et al., 2013) and wood waste left in the forest (Butarbutar et al., 2019; Holmes et al., 2002; Medjibe and Putz, 2012). After the first harvest, RIL retains more biomass in the forest than CL, thus reducing carbon emissions from selective logging (Medjibe et al., 2011; Pinard and Putz, 1996; Vidal et al., 2016; West et al., 2014). Additionally, RIL can allow fast biomass

recovery (Butarbutar et al., 2019; Lussetti et al., 2019; Putz et al., 2008b; Vidal et al., 2016; West et al., 2014). Furthermore, RIL may not alter species richness (Chaudhary et al., 2016) and has lower impacts on biodiversity compared to CL (Bicknell et al., 2014; Griscom et al., 2018). It is worthy to highlight that the extent of selective logging damage does not only depend on the techniques used, but also on logging intensity (Sist, 2000). Nonetheless, logging intensity, in terms of timber volume harvested ha^{-1}, is generally higher in conventional logging compared to reduced-impact logging (Martin et al., 2015).

Despite the ecological advantages associated with RIL, high-damaged CL practices have persisted in the tropics (Holmes et al., 2002; Medjibe and Putz, 2012; Putz et al., 2008a, 2000). The principal reason is increased-costs associated with the implementation of RIL protocols (Medjibe and Putz, 2012; Putz et al., 2000). Therefore, compensating forest companies for revenues foregone from implementing RIL techniques can help in adopting RIL (Medjibe and Putz, 2012; Putz and Romero, 2015).

1.3. Selective logging context in Cameroon

Cameroon is a major timber producer and exporter in Africa (ITTO, 2019). Between 1990 and 2015, its forest cover has declined at a rate of 1% per year and was estimated at 18.8 million hectares (ha) in 2015, approximately 40% of the country's territory (Keenan et al., 2015). Cameroonian forest law (1994) divides the forest area into permanent and non-permanent forest estates covering 85% and 15% of the forestland respectively (MINFOF and WRI, 2018). Permanent forest estate is devoted to sustainable timber production and biodiversity conservation, while the non-permanent forest estate includes forests that can potentially be allocated to other land-uses. Forest are owned by the state and logging authorizations can be granted in both forest estates. The state transfers its rights of forest management to logging

companies through logging concessions, under the control of its Ministry of Forestry and Wildlife. A logging concession has a maximum area of 200,000 ha and can include one or several forest management units (FMU). FMU are basic units of harvesting that are managed under selective logging over a minimum rotation period of 30 years fixed by the forest law. It is the responsibility of the concessionaires to carry out forest inventories, draw up forest management plans and implement them after their approval by the forest administration. To ensure continuous supply of timber, concessionaires split the area of FMU into 30 annual harvest blocks (or annual cutting areas), each to be logged once during a logging cycle. The aim is to give the logged forest time to replenish harvestable timber stock for the next logging event in 30 years, which is however unlikely to happen (Bonnell et al., 2011; Lennox et al., 2018).

Selective logging in Cameroon involves the harvest of few valuable timber trees exceeding a threshold stem diameter at breast height (i.e., at 1.3 m height) specified by the forest administration and adjustable by the concessionaire to achieve better recovery and sustainability (Cerutti et al., 2008). By specifying a minimum cutting diameter per species and a maximum harvestable timber volume per annual block, the forest administration attempts to regulate timber extraction rates. As logging companies are the ones deciding which key species to include in the management plan, management decisions are mainly based on their economic concerns (Cerutti et al., 2008). As common in central Africa, logging is highly selective with less than one tree being harvested per hectare, representing < 15 m^3 ha^{-1} (Sonwa et al., 2011). Assuming an annual wood production of 2.5 x 10^6 m^3 (MINFOF, 2017), more than 200,000 ha of forest are affected each year by selective logging in Cameroon. Despite the high tree species diversity in Cameroonian forest, few commercial timber species are overexploited, with five to six species accounting for $> 70\%$ of the total timber export (Cerutti et al., 2016). Cameroon wood market has shifted from Europe to Asia, with China (50%)

being the main importer of Cameroonian logs in 2018, followed by Vietnam (27%) (ITTO, 2019). Indeed, Asia buys a wider range of species than Europe and logging companies are not willing to bear high costs associated with producing certified products required by European markets (ITTO, 2019).

Selective logging also happens in unmanaged forests of the non-permanent forest estate, either with small logging permits (e.g., sale of standing wood, personal cutting permit) or illegally. Illegal logging remain a major concern in Cameroon (Hoare, 2015; Smith, 2004). To address this issue, Cameroon has signed the Forest Law Enforcement, Governance and Trade/ Voluntary Partnership Agreement (VPA/FLEGT) with the European Union. However, its implementation is still slow (Hoare, 2015). Despite its introduction in the 1994 forest law, forest management plans have not always guaranteed sustainable forest management and conventional selective logging remains dominant (Cerutti et al., 2008). RIL practices have increasingly been used due to voluntary forest certification and its third-party audits. The first FMU was certified in 2005 and to date, 44 FMU, covering an area of > 3 million ha, have been certified under different labels including Forest Stewardship Council and Timber Origin and Legality (MINFOF, 2017).

1.4. Objectives and hypotheses

In comparison to Amazonia, little is known on biogeochemical effects of selective logging in the Congo Basin forests. The research presented in this dissertation aimed to assess the impacts of two selective logging systems (conventional and reduced-impact logging) on forest soils in Cameroon.

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The extent of selective logging damage varies with logging techniques and intensity (Pinard and Putz, 1996; Sist, 2000). Therefore, the first objective of this research was to quantify the extent of disturbed area and the amount of nutrient exported in harvested timber by each logging system. We hypothesized that RIL will disturb less area and export less nutrients in comparison to CL.

Selective logging alter soil physical and biochemical characteristics, but the changes are variable among the disturbed strata: road, logging deck, skidding trail and felling gap (Olander et al., 2005). The second objective of this research was to assess changes in soil biochemical characteristics in CL and RIL forests. For this objective, we hypothesized that highly disturbed strata (roads, logging decks and skidding trails) will lose larger amounts of SOC and nutrient elements than the felling gaps when compared to the undisturbed reference area. Moreover, SOC and nutrient element losses in the disturbed area of RIL will be lower than CL.

Alteration of soil physical and chemical characteristics due to selective logging can result in localized changes in soil GHG fluxes (Keller et al., 2005; Yashiro et al., 2008). The objectives of our last study were to assess the changes in soil CO_2, N_2O and CH_4 fluxes resulting from each selective logging system; and determine spatial and temporal controlling factors of soil CO_2, N_2O and CH_4 fluxes in selectively logged forest. Therefore, we hypothesized that soil CO_2 fluxes will be higher whereas soil N_2O and CH_4 fluxes will be lower in the undisturbed reference area and felling gaps than in the highly disturbed strata. Additionally, soil bulk density and WFPS that control gas diffusivity will control soil GHG fluxes. Furthermore, soil CO_2 and CH_4 fluxes will be controlled by soil fertility (SOC, C:N ratio, ECEC…), and soil N_2O fluxes by N availability.

1.5.   Overview of the research area

1.5.1.   Location and experimental design

The study was carried out in the technical operational unit (TOU) Campo-Ma'an (2°10'–2°52' N, 9°50'–10°54' E), located in the south Region of Cameroon (Fig. 1.1). The TOU Campo-Ma'an rainforest covers an area of ~7710 km^2, with > 65% in the permanent forest estate including a national park and five logging concessions. The remaining area is under the non-permanent forest estate and includes agro-industrial plantations and the agroforestry zone. Selective logging occurs in the production forest (i.e., logging concessions) and in the agroforestry zone (i.e., either sales of standing timber, community forest or even without any logging permit around villages). For our study, we selected two forests logged with conventional (CL) and reduced-impact (RIL) logging protocols respectively. The CL forest is located within the non-permanent forest estate, close to a village and was logged under a local agreement between the village elders and a private operator. The conventional logger did not carry out a pre-logging forest inventory, but employed skilled workers for the logging operations. The RIL site is an annual harvest block of a logging concession (i.e., permanent forest estate) certified for sustainable forest management. The logging operations in that site followed the prescriptions of the approved forest management plan and were subject to third-party audits by the certification body. They were carried out by trained crews under the supervision of the internal auditors.
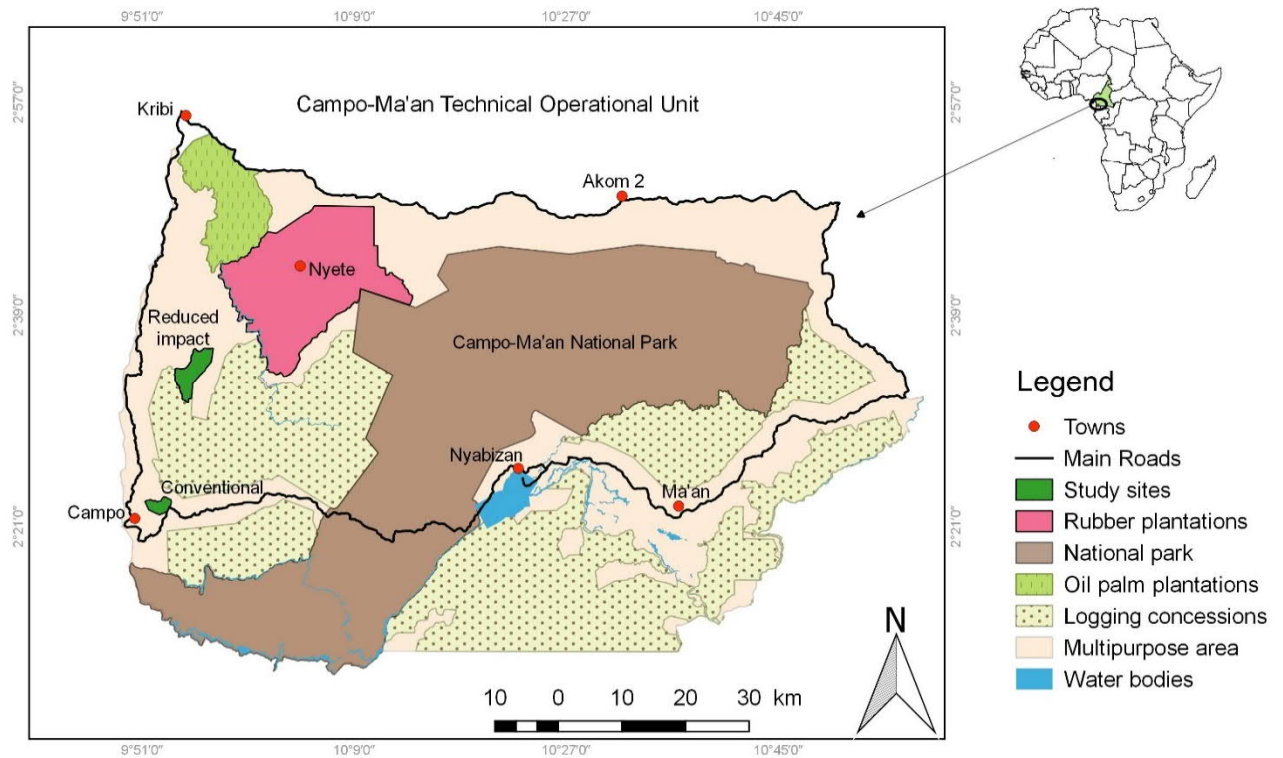
Fig. 1.1. Study site location in Campo-Ma'an Technical Operational Unit (South Cameroon).

The experimental design within each forest included the various strata resulting from selective logging (road, logging deck, skidding trail, felling gap and undisturbed area). Disturbed strata are related to different disturbance types: forest clearing and soil bulldozing in logging deck and road building, soil compaction on skidding trails and, canopy opening and waste accumulation on felling gaps (Fig. 1.2). Details on experimental and plot designs are given in Chapters 2 and 3.
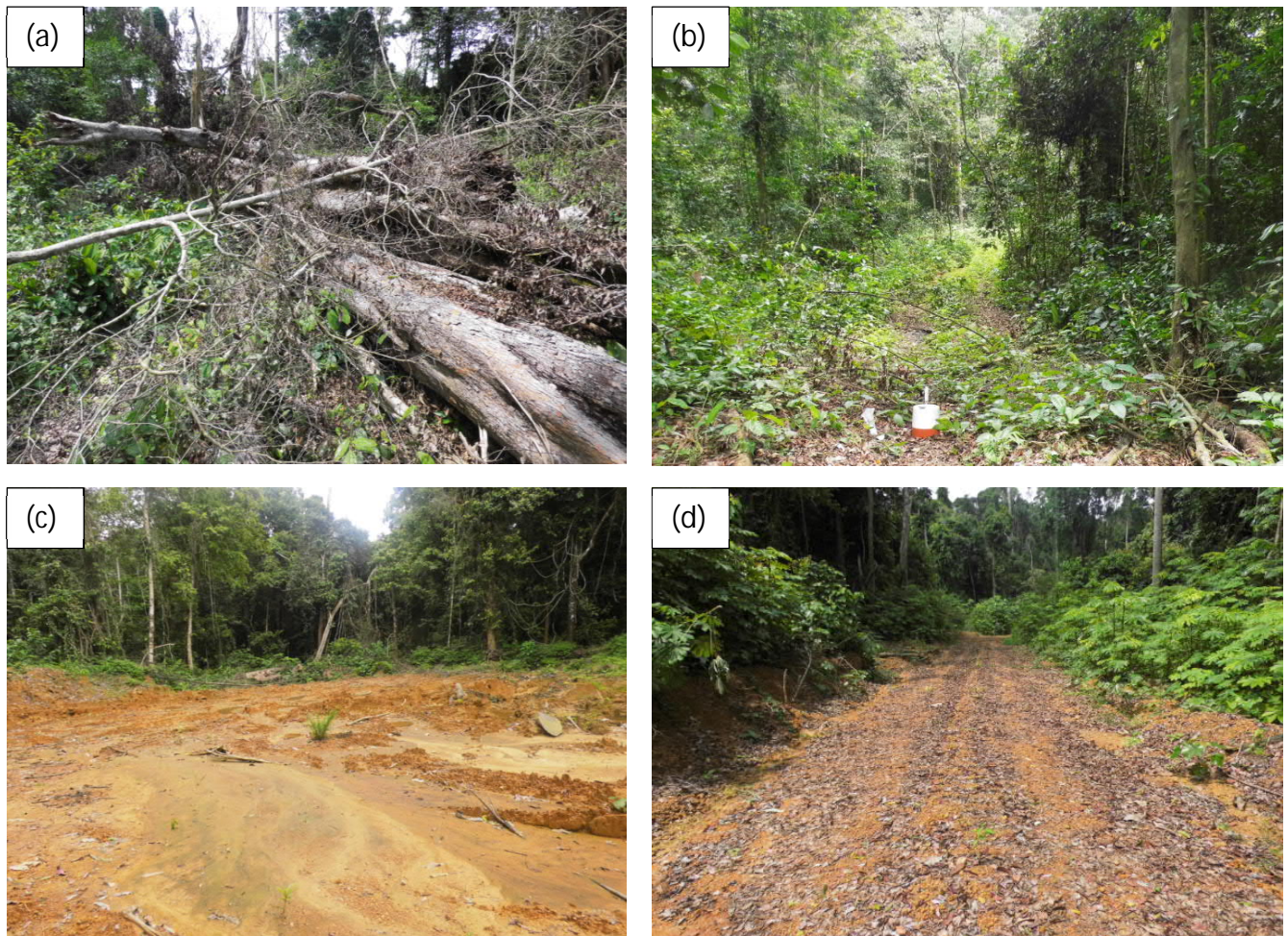
Fig. 1.2. Pictures of the disturbed ground strata 11 months after selective logging: (a) felling gap, (b) skidding trail, (c) logging deck, (d) road.

1.5.2. Climate and soil features

With at least 60 mm of average rainfall in all the months, the climate of the study area is tropical rainforest under Köppen climate classification, also referred to as equatorial climate (Peel et al., 2007). The average annual rainfall is 2690 mm and the mean annual temperature is 25.4°C (Climate-Data.org, 2018). There are two dry seasons with monthly rainfall < 150 mm (December to February and July to August) and two rainy seasons (March to June and September to November, Fig. 1.3).

Soil was sampled down to 50 cm in three depth intervals (0-10, 10-30 and 30-50 cm) in the undisturbed area within CL and RIL sites and analyzed for physical and biochemical characteristics (see Chapter 2). Soils of both sites have a very low effective cation exchange capacity, base saturation, pH and very high Al saturation, and are classified as Ferralsols (i.e., Oxisol) (Table 1.1). The soil texture is sandy-loam down to 50 cm depth.
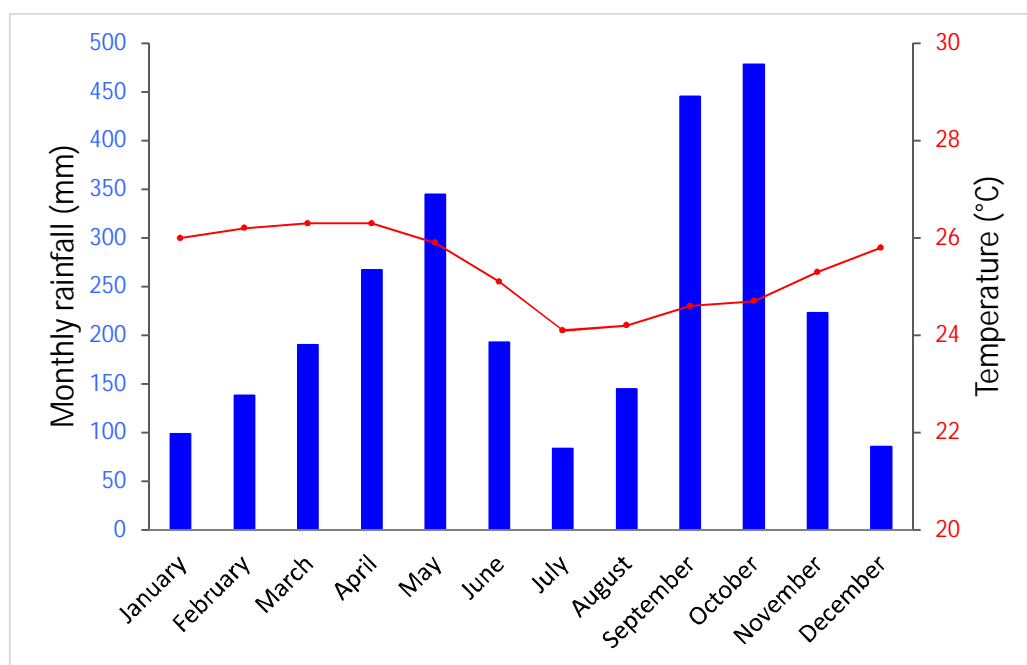
Fig. 1.3. Mean monthly rainfall and temperature in Campo, South Cameroon (Data source: Climate-Data.org, 2018).

Table 1.1

Soil characteristics (mean ± SE) at three depth intervals down to 50 cm in the CL and RIL forests.

Soil characteristics	Conventional logging forest			Reduced-impact logging forest		
	0-10 cm	10-30 cm	30-50 cm	0-10 cm	10-30 cm	30-50 cm
pH (1:2.5 H_2O)	3.9 ± 0.1	4.2 ± 0.1	4.3 ± 0.1	4.2 ± 0.1	4.5 ± 0.1	4.7 ± 0.1
Effective cation exchange capacity ($cmol_c$ kg^{-1})	2.5 ± 0.1	1.6 ± 0.1	1.6 ± 0.3	3.3 ± 0.2	2.2 ± 0.2	1.6 ± 0.1
Base saturation (%)	15.2 ± 1.9	14.3 ± 1.8	13.5 ± 3.5	20.1 ± 4.0	19.5 ± 2.4	17.7 ± 3.7
Aluminum saturation (%)	69.2 ± 2.1	75.6 ± 1.2	77.8 ± 2.5	65.2 ± 5.4	70.8 ± 2.8	73.6 ± 4.4
Texture: Sand (%)	55.0 ± 3.0	55.6 ± 2.4	55.5 ± 2.1	64.8 ± 3.0	64.0 ± 1.0	64.5 ± 1.5
Clay (%)	13.3 ± 0.5	12.8 ± 2.9	10.8 ± 1.7	12.3 ± 0.5	12.5 ± 3.5	14.3 ± 4.2

1.5.3. Vegetation structure

A tree inventory was carried out in the undisturbed area of the CL and RIL forest to give an overview of the vegetation structure in the research area. All stems with a diameter at breast height (DBH) ≥ 10 cm were identified and measured for DBH (i.e., stem diameter at 1.3 m above the ground) within four replicate plots (50 m × 50 m each) randomly selected at each site. The results of the inventory revealed similar vegetation characteristics between CL and RIL forests, although the species richness was higher in the RIL forest (Table 1.2). Nonetheless, both forests are highly species-diverse with Shannon diversity index > 3. In all plots, a total of 52 and 99 species belonging to 27 and 36 families were identified at the CL and RIL sites respectively. The most abundant family was Fabaceae (or Leguminosae) that accounted for more than 20% of the total trees.

Table 1.2

Vegetation characteristics (mean $\pm$ SE; $n = 4$ plots) of the undisturbed area within the forests logged by conventional selective logging and reduced-impact logging.

Vegetation characteristics	Conventional logging forest	Reduced-impact logging forest
Tree density (trees ha^{-1})	230 $\pm$ 14	243 $\pm$ 23
Shannon diversity index	3.1 $\pm$ 0.0[b]	3.6 $\pm$ 0.1[a]
Mean DBH (cm)	29.0 $\pm$ 0.6	29.3 $\pm$ 0.9
Basal area (m^2 ha^{-1})	23.6 $\pm$ 1.2	29.2 $\pm$ 2.5
Aboveground biomass [†] (Mg ha^{-1})	345 $\pm$ 35	471 $\pm$ 42
Aboveground carbon stock [‡] (Mg C ha^{-1})	161 $\pm$ 17	219 $\pm$ 19
Legume abundance (% tree)	23.5	20.7

Mean within a row followed by a different letters indicate significant differences between CL and RIL forests (Independent t-test at $P \leq 0.05$).

[†] Aboveground biomass $= \rho \times \exp(-1.499 + 2.148 \ln(DBH) + 0.207 (\ln(DBH))^2 - 0.0281 (\ln(DBH))^3$ (Chave et al., 2005); ρ is the wood density in g cm^{-3} (from Zanne et al., 2009); *DBH* in cm.

[‡] Aboveground carbon stock = 46.53% Aboveground biomass (Djomo et al., 2011).

Tree densities (Table 1.2) were within the range previously reported for the agroforestry zone and managed forests in the area of Campo-Ma'an (216–398 trees ha^{-1}; Djomo et al., 2011). Timber species that were logged in those forests (Table 2.1) represented 12 to 14% of the total tree population in RIL and CL respectively. The inverted J-shape diameter distribution of these forests (Fig. 1.4) reveals their very high potential for natural regeneration. Indeed, 43 to 53% of the population are small-sized trees (10–20 cm), which result in a low mean DBH (< 30 cm). There is a potential for high mortality of small trees, which limits the recruitment to larger diameter classes. Although selective logging may have influenced the aboveground

biomass (AGB) measured in our sites, our values were still at the upper end of the range of

pre-logging aboveground carbon stocks reported for forests in the same area (86–250 Mg C

ha^{-1}; Djomo et al., 2011). This highlights that biomass stocking in intact areas within selective

logging landscapes can reflect their pre-logging status (Putz et al., 2019).
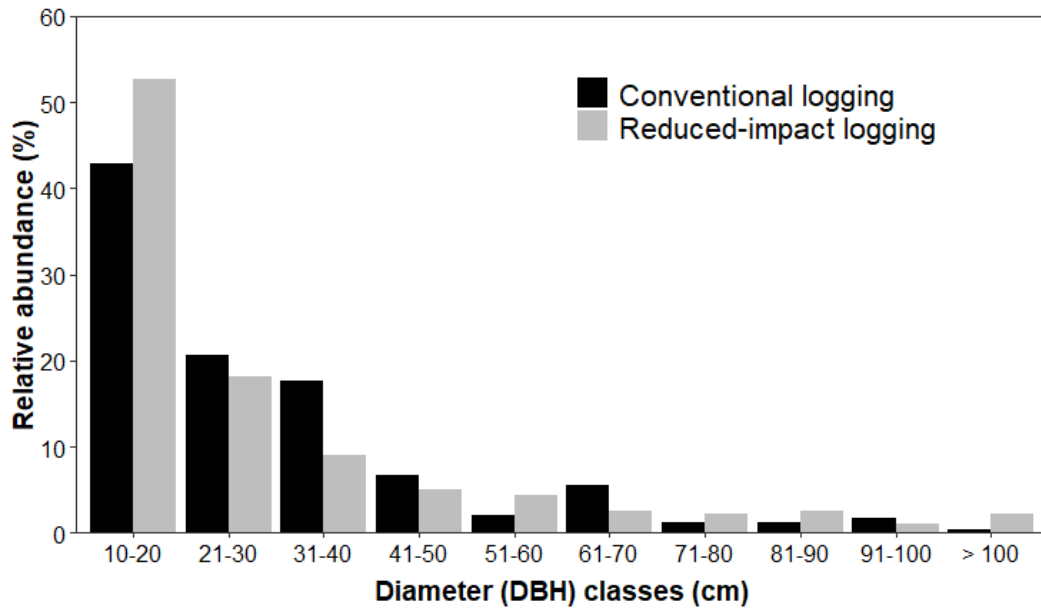
Fig. 1.4. Tree diameter distribution within the undisturbed area of conventional and reduced-impact logging forests.

1.6. References

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CHAPTER 2

CHANGES IN SOIL ORGANIC CARBON AND NUTRIENT STOCKS IN CONVENTIONAL SELECTIVE LOGGING VERSUS REDUCED-IMPACT LOGGING IN RAINFORESTS ON HIGHLY WEATHERED SOILS IN SOUTHERN CAMEROON

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2.1. Abstract

Although disturbances associated with selective logging can cause pronounced changes in soil characteristics and nutrient stocks, such information is very limited for highly weathered soils in Africa. We assessed the effects of reduced impact logging (RIL, with a 30-year rotation management plan) and conventional logging (CL, without a management plan) on physical and biochemical characteristics of Ferralsol soils that developed on pre-Cambrian rocks in rainforests of Cameroon. Five to seven months after the logging operations were completed, we mapped the CL and RIL sites and quantified the disturbed areas: felling gaps, skidding trails, logging decks and roads. We selected four replicate plots at each site that encompassed these four disturbed strata and an adjacent undisturbed area as the reference. At each disturbed stratum and reference area per plot, we took soil samples down to 50 cm, and quantified soil physical and biochemical characteristics. Nutrient exports with timber harvest were also quantified. The logging intensity was very low with removals of 0.2 and 0.3 tree per hectare, and the ground area disturbed accounted only 5.2% and 4.0% of the total area in CL and RIL, respectively. In terms of area disturbance for each harvested tree, CL had 753 m^2 $tree^{-1}$ more affected ground area than RIL. Roads and logging decks were the most affected by logging operations, where effective cation exchange capacity, soil organic carbon (SOC), total nitrogen (N), Bray-extractable phosphorus (P) and exchangeable aluminum decreased whereas pH, ^{15}N natural abundance and exchangeable manganese increased compared to the undisturbed reference area ($P < 0.01$–0.04). The disturbed area showed overall reductions of 21–29% in SOC, N and P stocks relative to the reference areas ($P = 0.02$–0.07). The amounts of C, N, P and base cations exported with harvested timber were only 0.4–5.9% of the changes in stocks of these elements in the disturbed strata. Nutrient reductions in the soil and exports through timber harvest were comparable between CL and RIL, after one logging event in this very low intensity logging systems. Our results suggest that unplanned operations together

with frequent re-logging inherent to CL can increase area damage and enhance changes in SOC and nutrients as opposed to RIL, which may affect the recovery of the succeeding vegetation.

Keywords: selective logging, ground disturbance, soil characteristic changes, nutrient export, Ferralsols, Cameroon.

2.2.   Introduction

Selective logging is one of the main drivers of forest degradation in developing countries, mainly through the construction of dense networks of access roads (Bell et al., 2012; Hosonuma et al., 2012; Kissinger et al., 2012). In many tropical forests, only the largest and highest quality trees of merchantable species are harvested, where only logs are extracted and the crown and woody debris are left on the forest floor to decompose (Asner et al., 2005). Harvest operations result in various disturbed strata (road, logging deck, skidding trail and felling gap) that are localized to small areas, making it difficult to monitor selective logging using satellite imagery (Asner et al., 2005). The extent of these disturbed strata is highly correlated to the logging intensity, expressed in terms of either timber volume or number of trees harvested per hectare (ha) (Durrieu de Madron et al., 2000; Pereira et al., 2002). Although the impact across the entire logged area may be minimal, some areas are quite severely disturbed (roads, logging decks and skidding trails) and experience soil compaction that can last for decades (Bol and Tokuchi, 2018; Keller et al., 2005; McNabb et al., 1997; Olander et al., 2005). Forest soil characteristics can respond differently to these various selective logging-related disturbances.

To date, most studies on biogeochemical effects of selective logging have been carried out in Amazonia, where up to 16 trees are harvested per hectare (McNabb et al., 1997; Olander

et al., 2005; Villela et al., 2006). Forests in Africa account for 25% of the total biomass carbon stock of tropical forests (Saatchi et al., 2011), of which approximately 30% are in industrial logging concessions (Laporte et al., 2007; Nasi et al., 2006) with logging intensities ranging between < 1 to 2 trees ha^{-1} (Brown et al., 2005; Durrieu de Madron et al., 2000; Jonkers and van Leersum, 2000; Medjibe et al., 2013). Previous studies on responses of African forests to selective logging disturbance mainly focused on stand structure, tree diversity, above-ground biomass, forest dynamics and wildlife populations (Cazzolla Gatti et al., 2015; Hall et al., 2003; Malcolm and Ray, 2000; Medjibe et al., 2013, 2011; Nzogang, 2009; Vaglio Laurin et al., 2016). The few studies that have assessed changes in soil organic carbon (SOC; Chiti et al., 2016; Vaglio Laurin et al., 2016) and nutrient status (Asase et al., 2014) were not spatially explicit.

Soil quality is crucial for sustainable management of African forests, as changes in biochemical properties of their nutrient-poor soils will affect their future productivity (Laurance et al., 1999). Cameroon is a major timber producer in Africa with an average annual production of 2.3 million cubic meters of wood (Cerutti et al., 2016). Indeed, more than seven million ha of its natural forests have been allocated for timber production, whereby the majority is selectively logged. There are two common selective logging methods utilized in Cameroon, namely conventional logging (CL) and reduced-impact logging (RIL). There is no proper planning of logging operations in CL, which are generally uncontrolled and carried out by untrained crews. Moreover, re-logging after short periods is a common practice of CL, driven by market demand and governance failure. On the other hand, RIL aims to reduce the negative impacts of selective logging on soils and residual stands through careful planning and control of logging operations and the use of improved measures such as directional tree felling techniques, road and skidding trail post-harvest closure (Putz et al., 2008). It has been shown that RIL can reduce the ground damage and above- and below-ground carbon losses

compared to CL, particularly in areas with high logging intensities (Keller et al., 2004; Pereira et al., 2002; Pinard and Putz, 1996). Selective logging has been shown to alter soil physical and chemical properties, resulting in the depletion of SOC, nitrogen (N) and phosphorus (P) stocks mainly in more disturbed areas (Bol and Tokuchi, 2018; Chiti et al., 2016; Durigan et al., 2017; McNabb et al., 1997; Olander et al., 2005; Vaglio Laurin et al., 2016). On the other hand, increases in soil pH, calcium (Ca) and magnesium (Mg) were reported in disturbed areas at moderate to high logging intensities (> 3 trees ha^{-1}) (McNabb et al., 1997; Olander et al., 2005). Nevertheless, some authors found little or no effect of selective logging on soil nutrient concentration (Asase et al., 2014; Villela et al., 2006). None of these studies compared the effects of CL and RIL on soil characteristics.

In this study, we investigated the effects of low intensity selective logging at two sites on heavily weathered soils in southern Cameroon. Specifically, our objectives were to (1) assess changes in soil biochemical characteristics in CL versus RIL, and (2) quantify the extent of disturbed area by each logging system. We designed our study such that each of the four replicate plots at each logging system covered the four disturbed strata (road, logging deck, skidding trail and felling gap) and an adjacent undisturbed area as the reference. We tested the following hypotheses: (1) in comparison to the undisturbed reference area, the roads and logging decks will lose larger amounts of SOC and nutrient elements than the less disturbed skidding trails and felling gaps; and (2) RIL will have less disturbed area, SOC and nutrient element losses in comparison to CL.

2.3. Materials and methods

2.3.1. Study area

The study was conducted in the Technical Operational Unit (TOU) Campo-Ma'an located in southern Cameroon (2°10'–2°52' N, 9°50'–10°54' E). It is a 7710-km^2 rainforest, which includes a protected area (national park), logging concessions, agro-industrial plantations and multipurpose area (Fig. 1.1). The study area has an equatorial climate with two dry seasons when monthly rainfall is less than 150 mm (December to February and July to August) and two rainy seasons (March to June and September to November). The average annual rainfall is 2690 mm and the mean annual temperature is 25.4°C (Climate-Data.org, 2018). Geologically, the area is situated on inferior pre-Cambrian basement rocks, made up of metamorphic micaschists, superior and inferior gneiss and undifferentiated gneiss (Gwanfogbe et al., 1983).

We selected two lowland forests (elevation range of 30–100 m above sea level) with a relatively flat topography, located approximately 40 km apart. Both forests were selectively logged using similar machinery, five to seven months prior to our study, where one site was conventionally logged and the other was logged using RIL protocols. The CL site was logged without a management plan and minimal advanced planning. The logger located the desired trees, which were then felled with little consideration to the surrounding trees, and the skidding trails were opened to find the felling gaps. In contrast, RIL was conducted with an approved forest management plan based on a 30-year logging cycle, and it was the first logging event. Management practices in RIL included inventory of trees, completed one year before logging, planning of skidding trails and roads with emphasis on minimizing their width and density, and the use of controlled tree felling by trained crews. The CL site is located in Mintom village (2°22'37"N, 9°52'28"E, Fig. 1.1), and was logged under a local agreement

between the village and a private operator. This forest was probably logged before, but according to the village elders, this was the first mechanical logging entry in at least 25 years. The CL logger chose to work with the local people who were previously employed and trained by other logging companies. The RIL site is located in a logging concession (2°33'08"N, 9°55'03"E, Fig. 1.1) owned by a commercial enterprise certified by the Forest Stewardship Council for sustainable forest management.

Both CL and RIL sites are on heavily weathered soils classified as Ferralsols with a sandy loam texture (Table 2.1). These soils have very low effective cation exchange capacity (ECEC), base saturation, pH and very high Al saturation (Table 2.2). The CL and RIL sites, based on the undisturbed reference plots (see 2.2.3 below), were similar in soil texture and element stocks in the soils (Tables 2.1 and 2.2). These suggest that both forests had comparable initial soil conditions prior to logging and any differences in their disturbed area (see 2.2.2 below) can be attributed to their logging systems. Both forests had also similar diameter at breast height (DBH) of trees, tree density, basal area, and tree species with the highest importance value index (IVI) (Table 2.1). These parameters were assessed through a vegetation survey in four replicate plots (50 m × 50 m each), selected randomly in the undisturbed area at each site. Within each replicate plot, all stems with ≥ 10 cm DBH (at 1.3 m above the ground or 0.3 m above the buttresses) were identified for species and measured using a diameter tape measure. The most dominant tree species were identified based on IVI as follows:

IVI = relative density + relative frequency + relative dominance (Curtis and McIntosh, 1951). For a given species, the relative density refers to its total number of individuals in the four plots, the relative frequency refers to its occurrence among the four plots, and the relative dominance refers to its total basal area in the four plots, all expressed as percentages of all species.

Table 2.1

Characteristics (mean ± SE, *n* = 4 plots) of the two forests logged by conventional selective logging and reduced-impact logging.

Site characteristics	Conventional logging	Reduced-impact logging
Soil texture [†]: Sand (%)	55.3 ± 2.4	64.4 ± 1.6
Silt (%)	32.7 ± 2.2	22.5 ± 4.0
Clay (%)	12.0 ± 1.9	13.1 ± 3.2
Tree density (N ha^{-1}) [‡]	230 ± 14	243 ± 23
Basal area (m^2 ha^{-1}) [‡]	23.6 ± 1.2	29.2 ± 2.4
DBH (cm) [‡]	29.0 ± 0.6	29.3 ± 0.9
Logging intensity (m^3 ha^{-1}) [§]	2.75	2.78
Dominant tree species (IVI) [¶]	*Sacoglottis gabonensis* (24)	*Sacoglottis gabonensis* (15)
	Tabernae montana crassa (18)	*Coula edulis* (12)
	Gilbertiodendron dewevrei (15)	*Strombosia pustulata* (10)
	Poga oleosa (12)	*Blighia welwitschii* (10)
	Hexalobus crispiflorus (12)	*Irvingia gabonensis* (10)
Most harvested tree species [††] and their relative proportion of the total harvested trees [¶] (%)	*Lophira alata* (56)	*Lophira alata* (39)
	Erythropleum ivorense (17)	*Erythropleum ivorense* (30)
	Guibourtia ehie (13)	*Dialium bipendensis* (16)
	Pterocarpus soyauxii (10)	*Pterocarpus soyauxii* (5)
	Piptadeniastrum africanum (4)	*Lovoa trichilioides* (3)

[†] Depth-weighted average of the sampled soil depths (0–10, 10–30 and 30–50 cm), determined in the undisturbed reference area.

[‡] Determine in the undisturbed reference area (four 50 m × 50 m replicate plots in each logging system) for trees with ≥ 10 cm diameter at breast height (DBH at 1.3 m above the ground or 0.3 m above the buttresses).

[§] Total over-bark timber volume divided by the logged forest area.

[¶] Top 5 species with the highest Importance Value Index [IVI = relative density + relative frequency + relative dominance (Curtis and McIntosh, 1951)].

[††] Top 5 species with the highest number of trees harvested.

¶ Relative proportion = $\dfrac{\text{Number of trees harvested for a species}}{\text{Total number of trees harvested in the site}} \times 100$

Table 2.2

Soil physical and biochemical characteristics (mean $\pm$ SE, $n = 4$ plots) in the top 50 cm [†] soil depth for the undisturbed reference area and disturbed strata within each logging system.

Logging system	Reference	Felling gap	Skidding trail	Logging deck	Road
Conventional					
Bulk density (g cm^{-3})	1.3 $\pm$ 0.1[a]	1.4 $\pm$ 0.1[a]	1.4 $\pm$ 0.1[a,A]	1.6 $\pm$ 0.1[a,A]	1.5 $\pm$ 0.1[a]
pH (1:2.5 H$_2$O)	4.2 $\pm$ 0.1[b,B]	4.2 $\pm$ 0.1[b,B]	4.4 $\pm$ 0.1[ab,B]	4.7 $\pm$ 0.1[a]	4.7 $\pm$ 0.1[a,B]
Effective cation exchange capacity (ECEC, cmol$_c$ kg^{-1})	1.8 $\pm$ 0.1[a,B]	1.6 $\pm$ 0.1[ab]	1.2 $\pm$ 0.0[bc,B]	1.1 $\pm$ 0.2[bc,B]	1.0 $\pm$ 0.1[c]
Base saturation (%)	14.2 $\pm$ 1.7[b]	12.8 $\pm$ 2.3[b,B]	13.0 $\pm$ 2.2[b,B]	22.9 $\pm$ 6.0[ab]	29.0 $\pm$ 6.5[a]
Aluminum saturation (%)	75.2 $\pm$ 0.6[a]	75.6 $\pm$ 1.7[a,A]	76.7 $\pm$ 1.7[a,A]	68.3 $\pm$ 5.3[ab]	63.0 $\pm$ 5.5[b]
^{15}N natural abundance (‰)	8.5 $\pm$ 0.3[a,A]	8.4 $\pm$ 0.2[a]	8.9 $\pm$ 0.1[a,A]	9.0 $\pm$ 0.3[a,A]	9.0 $\pm$ 0.2[a,A]
^{15}N enrichment factor (‰) [‡]	-1.3 $\pm$ 0.2[a]	-1.3 $\pm$ 0.2[a]	-1.6 $\pm$ 0.1[a]	-	-
C:N ratio	12.9 $\pm$ 0.7[a]	12.4 $\pm$ 0.6[a]	11.7 $\pm$ 0.3[a,B]	13.0 $\pm$ 0.5[a]	11.8 $\pm$ 0.9[a]
Soil organic C (kg m^{-2})	6.9 $\pm$ 0.9[a]	5.4 $\pm$ 0.6[ab]	4.9 $\pm$ 0.4[ab,B]	4.8 $\pm$ 0.5[ab]	3.9 $\pm$ 0.3[b]
Total N (g m^{-2})	534 $\pm$ 42[a]	432 $\pm$ 54[ab]	413 $\pm$ 44[ab]	373 $\pm$ 46[b]	346 $\pm$ 8[b]
Bray-extractable P (g m^{-2})	2.1 $\pm$ 0.3[a]	2.0 $\pm$ 0.2[a]	1.5 $\pm$ 0.1[a]	1.4 $\pm$ 0.3[a]	1.3 $\pm$ 0.3[a,A]
Mg (g m^{-2})	5.5 $\pm$ 0.9[a]	4.8 $\pm$ 1.3[a]	2.8 $\pm$ 0.4[a,B]	6.0 $\pm$ 2.1[a]	5.1 $\pm$ 0.6[a]
Ca (g m^{-2})	12.2 $\pm$ 2.3[a]	11.1 $\pm$ 2.5[a,B]	9.7 $\pm$ 1.5[a,B]	13.8 $\pm$ 2.4[a,B]	18.8 $\pm$ 2.4[a]
K (g m^{-2})	10.4 $\pm$ 1.6[a]	10.6 $\pm$ 3.1[a]	5.4 $\pm$ 1.1[a]	10.2 $\pm$ 4.6[a]	10.4 $\pm$ 1.2[a,A]
Na (g m^{-2})	4.5 $\pm$ 1.9[a]	1.5 $\pm$ 0.6[a]	2.1 $\pm$ 0.8[a]	2.0 $\pm$ 1.0[a]	1.2 $\pm$ 0.4[a]
Aluminum (g m^{-2})	73.6 $\pm$ 6.3[a]	65.4 $\pm$ 3.7[ab,A]	50.8 $\pm$ 1.7[abc]	42.5 $\pm$ 7.2[bc]	36.7 $\pm$ 7.3[c]
Fe (g m^{-2})	6.5 $\pm$ 1.1[a]	5.6 $\pm$ 1.3[a]	3.5 $\pm$ 0.7[a]	4.1 $\pm$ 1.9[a]	1.9 $\pm$ 0.7[a]
Mn (g m^{-2})	1.4 $\pm$ 0.4[bc]	0.8 $\pm$ 0.2[c]	1.7 $\pm$ 0.4[abc]	2.5 $\pm$ 0.5[ab,A]	3.1 $\pm$ 0.8[a,A]
H (g m^{-2})	0.8 $\pm$ 0.2[a]	0.8 $\pm$ 0.2[a]	0.5 $\pm$ 0.1[ab]	0.3 $\pm$ 0.1[b]	0.3 $\pm$ 0.1[b]

Reduced-impact

Bulk density (g cm^{-3})	1.2 ± 0.0^a	1.1 ± 0.1^a	$1.1 \pm 0.0^{a,B}$	$1.2 \pm 0.1^{a,B}$	1.4 ± 0.2^a
pH (1:2.5 H$_2$O)	$4.6 \pm 0.1^{b,A}$	$4.8 \pm 0.0^{ab,A}$	4.8 ± 0.2^{ab}	4.9 ± 0.1^{ab}	$5.0 \pm 0.0^{a,A}$
ECEC (cmol$_c$ kg^{-1})	$2.2 \pm 0.1^{a,A}$	1.6 ± 0.1^{ab}	$2.0 \pm 0.2^{a,A}$	$1.7 \pm 0.2^{ab,A}$	1.1 ± 0.1^b
Base saturation (%)	18.9 ± 3.0^a	$27.2 \pm 3.1^{a,A}$	$26.9 \pm 5.6^{a,A}$	27.5 ± 6.6^a	23.8 ± 1.3^a
Aluminum saturation (%)	70.8 ± 3.8^a	$65.0 \pm 3.3^{a,B}$	$63.7 \pm 5.0^{a,B}$	64.9 ± 5.9^a	69.1 ± 1.0^a
^{15}N natural abundance (‰)	$7.2 \pm 0.2^{b,B}$	8.2 ± 0.2^a	$7.2 \pm 0.4^{b,B}$	$7.4 \pm 0.2^{ab,B}$	$8.0 \pm 0.2^{a,B}$
^{15}N enrichment factor (‰) [‡]	-1.1 ± 0.1^a	-1.3 ± 0.0^a	-1.3 ± 0.2^a	-	-
C:N ratio	14.0 ± 0.4^a	12.3 ± 0.1^a	$14.3 \pm 0.6^{a,B}$	15.0 ± 1.5^a	12.8 ± 0.5^a
Soil organic C (kg m^{-2})	8.8 ± 0.9^a	6.4 ± 0.6^{ab}	8.4 ± 1.2^a	6.3 ± 1.1^{ab}	4.1 ± 0.4^b
Total N (g m^{-2})	620 ± 59^a	516 ± 45^{ab}	577 ± 73^a	411 ± 44^{bc}	321 ± 73^c
Bray-extractable P (g m^{-2})	1.7 ± 0.2^a	1.5 ± 0.2^a	1.5 ± 0.3^a	0.8 ± 0.2^{ab}	$0.5 \pm 0.1^{b,B}$
Mg (g m^{-2})	8.3 ± 1.7^a	8.5 ± 2.6^a	$10.8 \pm 2.0^{a,A}$	7.2 ± 1.4^a	4.7 ± 0.4^a
Ca (g m^{-2})	24.4 ± 5.8^a	$26.0 \pm 3.9^{a,A}$	$35.4 \pm 7.7^{a,A}$	$31.0 \pm 6.0^{a,A}$	17.9 ± 1.0^a
K (g m^{-2})	13.1 ± 1.4^a	20.7 ± 5.9^a	14.1 ± 4.3^a	14.4 ± 2.2^a	$6.5 \pm 0.7^{a,B}$
Na (g m^{-2})	2.9 ± 0.8^a	2.1 ± 0.5^a	3.7 ± 0.9^a	1.3 ± 0.3^a	1.2 ± 1.0^a
Aluminum (g m^{-2})	78.4 ± 5.2^a	$52.8 \pm 1.2^{ab,B}$	64.9 ± 9.9^{ab}	58.0 ± 9.4^{ab}	40.4 ± 4.6^b
Fe (g m^{-2})	10.2 ± 1.7^a	5.6 ± 0.7^a	11.8 ± 4.6^a	6.7 ± 1.9^a	2.7 ± 0.5^a
Mn (g m^{-2})	1.0 ± 0.3^a	1.2 ± 0.3^a	1.7 ± 0.4^a	$1.3 \pm 0.2^{a,B}$	$1.2 \pm 0.2^{a,B}$
H (g m^{-2})	0.8 ± 0.1^a	0.4 ± 0.1^b	0.6 ± 0.2^{ab}	0.4 ± 0.1^b	0.3 ± 0.0^b

Means within a row followed by different lowercase letters indicate significant differences among strata within each logging system and uppercase letters indicate significant differences between logging systems within a stratum (ANOVA with Fisher's LSD test at $P \leq 0.05$ or marginal value of $P = 0.07$ for total N).

[†] Element stocks are the sum of the sampled soil depths (0–10, 10–30 and 30–50 cm) and all the rests are depth-weighted average, calculated for each replicate plot.

[‡] ^{15}N enrichment factor was not determined for logging deck and road because the calculation (see *Methods*) requires the litter layer (absent for these strata) as an input.

2.3.2. Area estimation of ground disturbance

The disturbed strata, namely the roads, logging decks, skidding trails and felling gaps of both CL (715 ha) and RIL (2350 ha) were mapped using a handheld global positioning system (GPS map 62s, Garmin Ltd., Southampton, Hampshire, United Kingdom). Roads and logging decks were bulldozed for temporary storage of logs, their loading on trucks and their off-forest transport. The skidding trails were used by the bulldozer to pull out the tree trunks to the logging decks. In felling gaps, the forest floor was covered by coarse and woody debris from felled trees. Lengths of roads and skidding trails were estimated in a geographic information system (GIS; QGIS version 3.4.2, Open Source Geospatial Foundation, Chicago, USA) whereas their widths were measured in the field at 25- to 50-m intervals using a tape measure. The total areas of the roads and skidding trails were calculated as the product of total length and averaged width. Length and width of all logging decks and every fourth felling gaps were measured in the field and their area estimated according to their shape. The average area of measured felling gaps was multiplied by the total number of felling gaps to estimate their total area coverage. Ground area disturbance was calculated for each stratum, and expressed both as disturbed area per harvested tree (i.e., disturbed area per ha divided by the logging intensity) and as percentage disturbed area of the total logged forest (Table S2.1).

2.3.3. Sampling design for soil and litter and analysis

Within CL and RIL sites, we established four replicate plots centered around four randomly selected logging decks, and each replicate plot encompassed the four disturbed strata and an undisturbed reference area (Fig. 2.1). Centered on each logging deck, we selected a 40-m road section, a 40-m skidding trail section and one felling gap. Within each replicate plot but

located 50 m away from any disturbed areas, we also sampled a 20-m × 20-m area of undisturbed forest (hereafter called the reference stratum). With this approach, we assumed that soil characteristics of the reference and disturbed strata were similar within a replicate plot prior to logging.

Soil and litter sampling was conducted between July and September 2016. At each stratum of each replicate plot, soil samples were collected from 12 randomly selected sampling points (with distances between 2 m and 10 m) in order to capture the spatial variability within each stratum per plot (Allen et al., 2016). For roads, logging decks and skidding trails, these 12 sampling points covered the breadth of these strata. At each sampling point, soil samples were taken using a soil auger from the top 50 cm at three intervals (0-10 cm, 10-30 cm, and 30-50 cm), where changes in soil biochemical properties due to land-use change mostly occurs (Goebes et al., 2019; van Straaten et al., 2015). For each stratum of each replicate plot, soils from the same depth intervals were mixed thoroughly into one composite sample. In total, 120 composite soil samples were collected (two logging systems × four replicate plots × five strata × three depth intervals). Additionally, intact soil cores (8 cm diameter and 5 cm length, 251 cm^3) were taken at the center of the three depth intervals from one soil pit per stratum of each replicate plot to determine the soil bulk density. As some soil cores contained stones or rock fragments, the bulk density was corrected for stone content. Leaf litter samples were collected, using a 20 cm × 20 cm frame, from three sampling points within the felling gap, skidding trail and reference area (not applicable for the road and logging deck), and composited for each stratum of each replicate plot. In total, 24 composite leaf litter samples were collected.

The composited soil samples were air-dried, 2-mm sieved, transported by air to Germany and dried again at 40°C prior to analysis. These were analyzed for texture, pH, extractable P, ECEC, SOC, total N and ^{15}N natural abundance. Soil texture was determined

for the reference area using pipette method after removal of iron oxide and organic matter (Kroetsch and Wang, 2008). Soil pH was measured from a 1:2.5 soil-to-distilled water ratio. Extractable P was determined using the Bray 2 method, which is commonly used for acidic, highly weathered soils in the tropics, and the extracted P was analyzed using inductively coupled plasma – atomic emission spectrometer (ICP-AES; iCAP 6300 Duo VIEW ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). The ECEC was determined by percolation with unbuffered 1 mol/L NH_4Cl and the exchanged cation concentrations (Mg, Ca, K, Na, Al, Fe, Mn) in percolates were measured using the ICP-AES described above. Base and Al saturations were calculated, respectively, as the percentage exchangeable bases (Mg, Ca, K and Na) and Al on ECEC. Subsamples of the composited soil samples and 60°C-dried composited litter samples were finely ground for measurement of total organic C, total N (using a CN analyser; Vario EL Cube, Elementar Analysis Systems GmbH, Hanau, Germany) and ^{15}N natural abundance signature (using isotope ratio mass spectrometry; Delta Plus, Finnigan MAT, Bremen, Germany).

For quantifying changes in SOC and nutrient elements between logging systems or between disturbed strata and an undisturbed reference area, it is important to use an equal amount of soil mass for the conversion of element content per soil-mass basis to an area basis. This is commonly done by adjusting the soil depth or, similarly, by using the soil bulk density of the undisturbed reference area. This is to avoid the confounding effects of possible differences in soil masses within a specific depth (e.g., de Blécourt et al., 2013; de Koning et al., 2003; van Straaten et al., 2015; Veldkamp, 1994). We reported the element stocks for the top 50 cm. For soil characteristics other than element stocks, these were weighted averages of the three depth intervals. For each replicate plot, we calculated the overall changes in SOC and nutrient stocks in the disturbed strata compared to the reference area; the stocks in the

disturbed strata were weighted by the strata's respective areal coverage (Table S2.1) and subtracted from the stocks in the undisturbed reference area.

For the strata where leaf litter was found (reference area, felling gap and skidding trail), we determined the ^{15}N natural abundance enrichment factor (ε) for the entire 50-cm depth, which is the slope of the line passing through the origin and the enrichment factors of each depth interval (ε_n). The enrichment factor at a certain depth (ε_n) was calculated using the Rayleigh equation (Baldos et al., 2015; Mariotti et al., 1981):

$\varepsilon_n\ (‰) = (d_n - d_0)\,/\,ln\ (\,f\,),$

where d_n and d_0 are the ^{15}N natural abundance signatures of a particular depth interval and the substrate input (or the leaf litter), respectively, and f is the remaining fraction of the total N (N concentration at a certain depth divided by the N concentration of the litter layer).
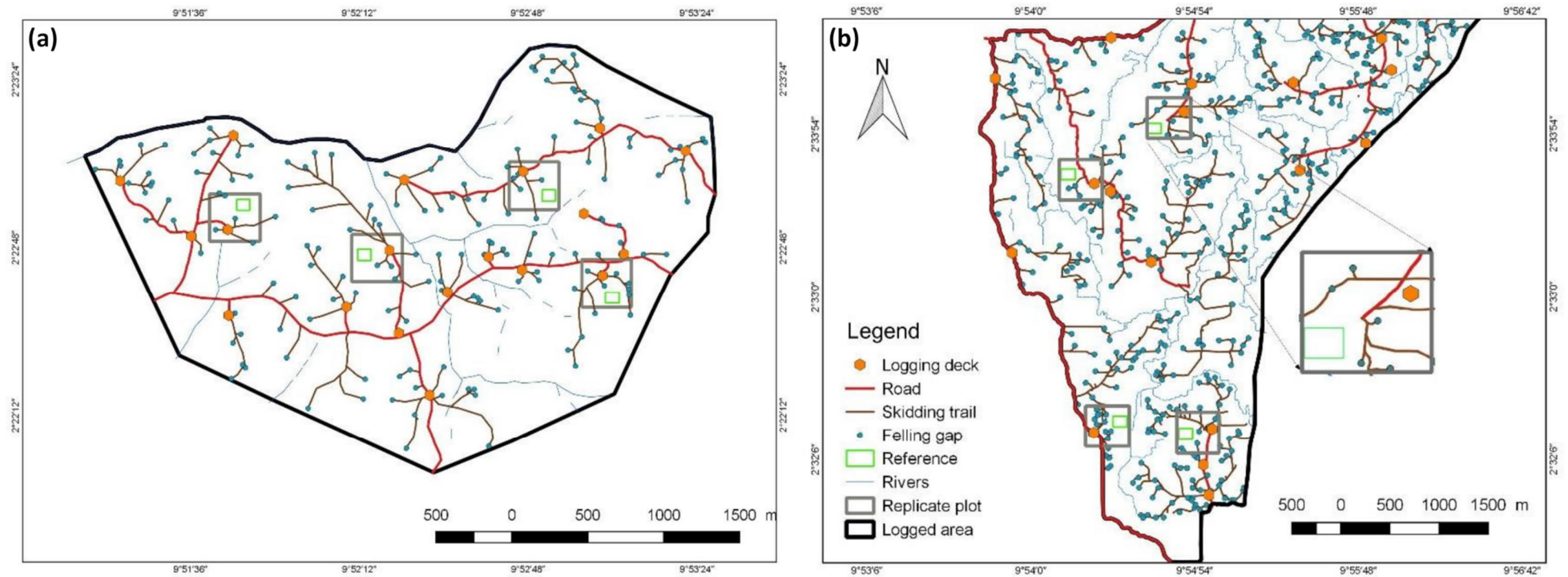
Fig. 2.1. Area surveyed and replicate plots, each with the disturbance strata (road, logging deck, skidding trail and felling gap), in (a) conventional selective logging (715 ha) and (b) reduced-impact logging (2350 ha). Each of the four replicate plots had a corresponding undisturbed reference area (20 m × 20 m), which was separated by at least 50 m distance from any disturbed stratum.

2.3.4. Estimation of nutrient export by timber harvest

The merchantable timber volumes, number of trees and species harvested in RIL were taken from the logging company data (Table 2.1). In CL, where there was no existing record, the timber volumes, number of trees and species were determined from all the stumps left in the field; timber volumes were estimated by taking two cross-sectional diameters on each stump. Stump diameter was converted to merchantable volume using species-specific regression equations developed from data of at least 25 timbers per species, which were collected from several active logging sites in southern Cameroon.

To estimate the amount of nutrients exported in the harvested timber, bole section samples including the three stem compartments (bark, sapwood and heartwood) were collected from three individuals per logged species (16 species in total for the two logging systems belonging to seven families). For each stem compartment, two sets of samples were taken to determine the specific gravity and the element concentration. The first set of bark or wood samples was measured for volume using suspension technique based on Archimedes' principle (Hughes, 2005), and then oven-dried at 105°C to constant weight for dry mass measurement (ASTM, 2007). The second set of bark or wood samples was oven–dried at 60°C and finely ground for total C, N and element concentration (Al, Ca, Fe, K, Mg, Mn, Na, P and S) analysis. Total C and N were measured using the same CN analyzer mentioned above. For total element concentrations, pressure digestion in concentrated HNO_3 was used and the digests were analyzed using the ICP-AES described above.

Specific gravity of a bark or wood sample was calculated as the ratio of the oven-dry mass to its fresh volume. Carbon and nutrient concentrations were converted into amounts of nutrients exported by multiplying the concentration of each element with the harvested timber mass (i.e., volume × wood specific gravity), weighted by the volume share of the stem

compartments. The volume share of the three stem compartments was estimated from 15 timbers per logged species (DBH > 50 cm) on which we determined the bark, sapwood and heartwood volumes as follows:

Volume share $_{bark}$ (%) = $((V_1 - V_2)/V_1) \times 100$

Volume share $_{sapwood}$ (%) = $((V_2 - V_3)/V_1) \times 100$

Volume share $_{heartwood}$ (%) = $(V_3/V_1) \times 100$

where V_1 = over bark volume; V_2 = under bark volume; V_3 = under sapwood volume.

Nutrient export was calculated per species (Table S2.2) and added up to determine the nutrient exported in harvested timber per logging system.

2.3.5. Statistical analyses

All variables were first checked for normal distribution (Shapiro-Wilk's test) and homogeneity of variance (Levene's test), and if these assumptions were not fulfilled the data were either log- or square root-transformed (i.e., N, Ca, and H). One-way ANOVA with Fisher's least significant difference test was used to evaluate the differences between strata or logging systems for variables with normal distribution and homogenous variance. When the assumptions were still not met after transformation, Kruskal-Wallis H test with multiple comparison extension was used (i.e., pH, C:N ratio and ^{15}N enrichment factor). We used the significant level at $P \leq 0.05$, and for a few soil characteristics (i.e., base and Al saturations in CL) we specified a marginal significance at $P = 0.07$. To test our first hypothesis, we compared the SOC and nutrients stocks among strata (road, logging deck, skidding trail, felling gap and reference area) within each logging system. For the second hypothesis, we compared the two logging systems for each stratum as well as the overall change in the disturbed strata (area-weighted of the four strata) from that of the undisturbed reference area (i.e., actual change =

disturbed strata – reference, or relative change = (disturbed strata – reference) ∕ reference).

Statistical analyses were conducted using R version 3.5.1 (R Core Team, 2018).

2.4. Results

2.4.1. Logging intensity and ground disturbance

Five species were harvested in CL compared to 14 in RIL, with three of the most harvested species being the same for both logging systems (Table 2.1). The top five harvested species, however, were not the species with the highest IVI (Table 2.1). Thus, the logging intensities were very low in both logging systems, in terms of harvested timber volume (Table 2.1) and in number of harvested trees (0.23 and 0.28 tree ha^{-1} for CL and RIL, respectively). Although the logging intensities were comparable between systems, the total ground area disturbed per harvested tree in RIL was 34% lower (equivalent to 753 m^2 tree^{-1}) than in CL (Fig. 2.2). Skidding trails and roads accounted for 75% of the disturbed area in both sites. Logging decks occupied the lowest fraction of the disturbed area and their density in CL was four times higher than RIL (Table S2.1). The area covered by felling gaps was also 22% lower in RIL (357 m^2 tree^{-1}) than in CL (460 m^2 tree^{-1}) (Fig. 2.2).
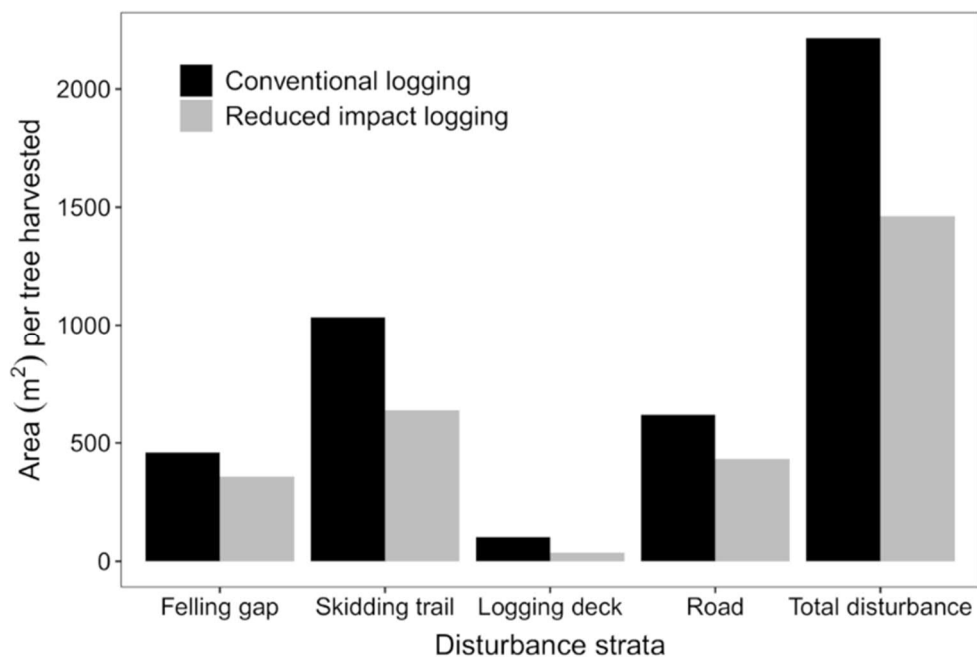
Fig. 2.2. Ground area disturbed (m^2) per tree harvested in conventional selective logging (715 ha with 167 harvested trees) and reduced-impact logging (2350 ha with 647 harvested trees).

2.4.2. Changes in soil characteristics and nutrient export by timber harvest

Soil physical (texture, bulk density) and biochemical characteristics (base and Al saturations and element stocks) in the reference areas of CL and RIL broadly did not differ (all $P > 0.05$; Tables 2.1 and 2.2). Exceptions were the lower soil pH and ECEC ($P < 0.01$–0.05) and higher ^{15}N natural abundance ($P = 0.01$) in the reference area of CL than that of RIL (Table 2.2).

      Among the disturbed strata in the CL, the roads and logging decks showed the largest changes in soil characteristics compared to the reference area. Specifically, roads and sometimes logging decks showed higher soil pH (or lower exchangeable H), higher base saturation and exchangeable Mn ($P < 0.01$–0.06), and lower ECEC, exchangeable Al (or Al saturation), SOC and total N stocks ($P < 0.01$–0.07) compared to the reference area (Table 2.2). The skidding trails showed intermediate changes in these characteristics whereas the felling gaps showed comparable values to the reference area (Table 2.2). In the RIL, similar

trends among the disturbed strata were observed except for base and Al saturations and exchangeable Mn stocks for which there were no differences detected among strata. Additionally, compared to the reference area, soil ^{15}N natural abundance was higher ($P = 0.02$) in the roads and felling gaps and extractable P was lower ($P < 0.01$) in the roads (Table 2.2). Most of these changes in soil characteristics occurred in the roads, while the other disturbed strata showed intermediate changes or comparable values to the reference area (Table 2.2).

The overall changes in area-weighted average of disturbed strata relative to the undisturbed reference area were comparable between CL and RIL for all soil characteristics (all $P > 0.3$; Fig. 2.3). The decreases in disturbed strata in SOC, N and P stocks, either in actual (Fig. 2.3a) or relative values (Fig. 2.3b), ranged from 21–29% of the reference areas.

The amounts of C, N, P and base cations exported with the harvested timber were considerably lower (Table 2.3) than the decreases in soil nutrient stocks at the disturbed strata (Fig. 2.3a). Nutrients exported by timber harvest represented only 0.4–5.9% of the changes in soil nutrient stocks at the disturbed strata of CL and 0.4–5.6% at the RIL. Average nutrient concentrations were higher in the bark than in the wood, and despite its low proportion in the harvested timber volume, bark accounted for 13–23% of nutrients exported (Table 2.3). As the main timber constituent, heartwood accounted for the highest proportion of C and nutrients exported, except for P that was highest in sapwood.
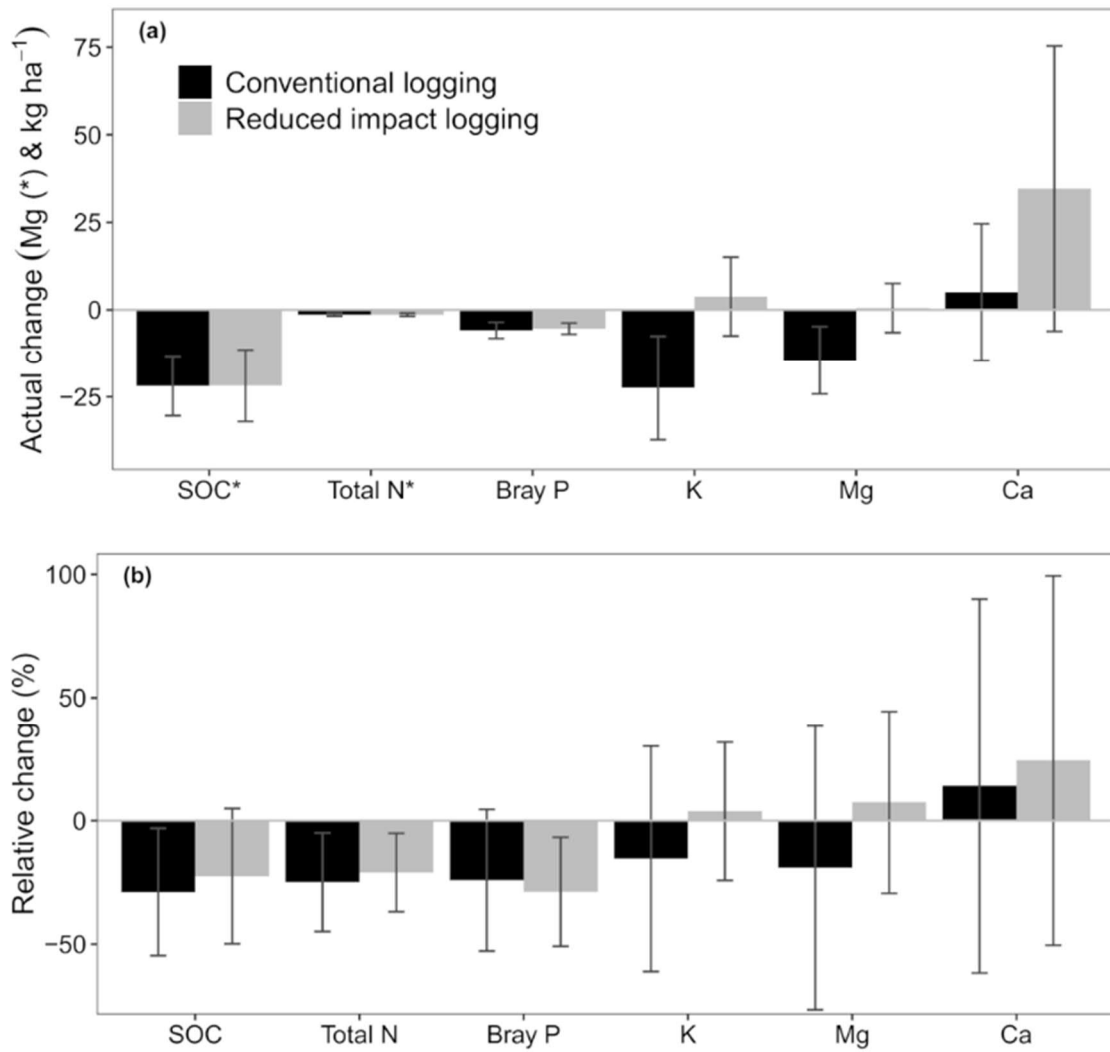
Fig. 2.3. Changes in soil organic C and macronutrients in the top 50 cm in conventional selective logging and reduced-impact logging: (a) actual change (area-weighted average of disturbed strata – reference; mean ± SE, $n = 4$ plots), and (b) relative change [(area-weighted average of disturbed strata – reference)/reference × 100; mean ± 95% confidence interval based on Student's *T* distribution, n = 4 plots].

Table 2.3

Contributions of the stem compartments to the volume of harvested timber, timber element concentrations (mean $\pm$ SE, n = 48 [16 species[†] $\times$ 3 individuals per species]), and exported elements from timber harvest.

Compartment	Volume share (%)	C	N	P	K	Mg	Ca
		Element concentration (g kg^{-1}$_{wood\ dry\ mass}$)					
Bark	5.5 $\pm$ 0.2	514.9 $\pm$ 3.8	10.5 $\pm$ 0.7	0.4 $\pm$ 0.0	3.5 $\pm$ 0.2	0.8 $\pm$ 0.1	10.3 $\pm$ 1.0
Sapwood	21.4 $\pm$ 0.8	529.8 $\pm$ 1.5	3.5 $\pm$ 0.2	0.2 $\pm$ 0.0	1.6 $\pm$ 0.1	0.3 $\pm$ 0.0	2.0 $\pm$ 0.2
Heartwood	73.1 $\pm$ 0.8	539.0 $\pm$ 1.6	2.9 $\pm$ 0.3	0.02 $\pm$ 0.0	0.5 $\pm$ 0.1	0.3 $\pm$ 0.1	1.8 $\pm$ 0.2
Entire log		535.7 $\pm$ 1.7	3.4 $\pm$ 0.3	0.1 $\pm$ 0.0	0.9 $\pm$ 0.1	0.3 $\pm$ 0.01	2.3 $\pm$ 0.2
Logging system		Nutrients exported in harvested timber (kg ha^{-1}) [‡]					
Conventional		1289.1	6.7	0.2	1.3	0.5	5.5
Reduced-impact		1227.4	6.2	0.2	1.4	0.6	4.3

[†] *Afzelia bipindensis, Afzelia pachyloba, Dialium bipendensis, Entandrophragma utile, Erythropleum ivorense, Guibourtia ehie, Khaya grandifoliola, Klainedoxa gabonensis, Lophira alata, Lovoa trichilioides, Mitragyna ciliata, Nauclea diderrichii, Piptadeniastrum africanum, Pterocarpus soyauxii, Pycnanthus angolensis, Tetraberlinia bifoliolata*

[‡] Nutrient export $= \sum_{i=1}^{n} \left(\text{harvested timber in kg}_{\text{dry mass}} \text{ ha}^{-1} \times \text{element concentration in kg kg}^{-1}_{\text{dry mass}}, \text{weighted by stem compartments} \right) \text{species}_i$

n = 5 and 14 species for conventional and reduced-impact logging respectively.

2.5. Discussion

2.5.1. Logging intensity and extent of ground disturbance

Logging intensity at our sites (< 0.3 trees ha^{-1}) was at a lower bound of those reported for Central Africa (0.3–1 tree ha^{-1}; Jonkers and van Leersum, 2000; Medjibe et al., 2011; Parren and Bongers, 2001). These values were, however, very low compared to the values reported for South America and Southeast Asia (ranging from 3 to 27 trees ha^{-1}; Asner et al., 2004; Bertault and Sist, 1997; Imai et al., 2012; Pinard et al., 2000; Saiful and Latiff, 2014; Sist, 2000; van der Hout, 2000). The low density of merchantable species at our sites (Table 2.1) explained the low logging intensity. Indeed, the most commonly harvested species were not the species with the highest IVI, indicating that the species considered merchantable were relatively few. The promotion of lesser-known species and diversification of the Cameroonian timber market, especially the increase of timber exports to China where a broader range of species is accepted than in European markets, can lead to an increase in the number of logged species and in turn in the logging intensity (Putzel et al., 2011; Tieguhong et al., 2015). This potential increase in timber demand for new species can also lead to re-logging after only a short period, which is more likely to happen in CL system.

The extent of ground disturbance at our sites (Table S2.1) was comparable to that observed in the Congo Basin forests with similar logging intensity (< 0.4 tree ha^{-1}) (Jonkers and van Leersum, 2000; Medjibe et al., 2013) but was much lower than those in Amazonian and Southeast Asian forests with higher logging intensities, especially when CL was utilized (Fig. S2.2; Asner et al., 2004; Holmes et al., 2002; Johns et al., 1996; Pereira et al., 2002; Pinard et al., 2000; Uhl and Vieira, 1989; van der Hout, 2000; Veríssimo et al., 1995). At our sites, the extent of mechanical ground disturbance (roads + logging decks + skidding trails) was only slightly higher in CL (4%) compared to RIL (3%), in contrast to previous studies

(Fig. S2.2) that show higher disturbance in CL. The relatively low ground disturbance at our CL site was related to the logging experience of workers employed by the CL operator. Using the skills of former employees of logging companies, minimal roads and skidding trails were opened in order to reduce the logging costs. This resulted in an alignment of skidding trails (Fig. 2.1) that is not common for CL practices (van der Hout and van Leersum, 2000). Nevertheless, one tree harvested in RIL disturbed 34% less area than in CL (Fig. 2.2), supporting previous comparative studies where RIL is successful in reducing ground disturbance by more than 50% compared to CL (Pereira et al., 2002; van der Hout, 2000).

The RIL guidelines prescribe various measures to reduce the logging damage (Putz et al., 2008) and our results highlight two of them, namely controlled felling and careful planning of skidding trails and logging decks prior to logging. The use of controlled felling techniques, with the aim of minimizing damage to adjacent trees, resulted in lowering the felling gap area per harvested tree by 22% in RIL compared to the CL (Fig. 2.2). However, the average felling gap area (Fig. 2.2) was at the upper bound of those reported for RIL in Brazil (166–355 m^2 $tree^{-1}$) with higher logging intensities (3.5–5.6 trees ha^{-1}) (Johns et al., 1996; Keller et al., 2005). The careful planning of logging deck locations at our RIL site, which takes into account the number of logs to be skidded to each of them, resulted in four times less logging decks than in the CL site (Table S2.1).

2.5.2. Soil characteristics in the reference area of the CL and RIL

The SOC, total N, extractable P and exchangeable K stocks in the top 50-cm depth of the reference areas in the CL and RIL sites were within the range of values reported for Ferralsol soils (or Oxisols) under tropical forests in Africa (Batjes, 2008; Chiti et al., 2016, 2014; Kanmegne, 2004; Onyekwelu et al., 2008; van Straaten et al., 2015). However, exchangeable

Mg and Ca stocks at our sites were lower compared to observations made in the same area (Kanmegne, 2004). Although the soil ^{15}N natural abundance at our sites was within the range reported for Ferralsol soils under lowland forests in Panama, our ECEC was well below those from Panamanian sites (Matson et al., 2017). The low ECEC at our sites could be due to the low clay content (Table 2.1).

Anecdotal evidence at our CL site suggests that this forest was probably logged before, although this was the first mechanical logging entry in at least the past 25 years. Those previous logging activities may be the cause for the reference areas in the CL site to have lower pH, lower ECEC, and higher ^{15}N natural abundance than in the RIL site (Table 2.1). Higher soil ^{15}N natural abundance could indicate a more decomposed soil organic matter (Baldos et al., 2015; Hassler et al., 2015; Sotta et al., 2008), possibly due to those previous disturbance, as well as a diminished contribution of organic matter to the pH-dependent ECEC, which is the main source of cation exchange in highly weathered Ferralsol soils. Nevertheless, the time from those previous disturbances in the CL site may have allowed sufficient recovery for the element stocks in the soil as none of these stocks differed ($P > 0.05$) from the reference area in the RIL site (Table 2.2).

2.5.3. Effects of selective logging

Selective logging, independent of logging methods used, affected a wide range of soil properties. Although some previous studies reported no effect of selective logging on soil physico-chemical properties and nutrient status (Asase et al., 2014; Villela et al., 2006), their contrasting results compared to our findings can be associated with their research design that failed to consider the spatial pattern of selective logging disturbances. By having an undisturbed reference area per replicate plot as the baseline for comparison, we were able to

detect changes in soil biochemical properties across a gradient of damage intensity from felling gaps that received a pulse input of fresh litter and woody debris to roads that underwent heavy compaction and topsoil removal. The most impacted areas were roads and to some extent logging decks, where we observed decreases in ECEC, SOC, total N, extractable P and exchangeable Al and increases in pH, soil ^{15}N natural abundance and exchangeable Mn compared to the reference areas (Table 2.2). Similar impacts of selective logging on soil properties were observed in clay-dominated Ferralsols in Amazonia (McNabb et al., 1997; Olander et al., 2005), these studies however considered only the surface soil layers (0–5 cm; 0–10 cm, respectively). Additionally, the results of our study highlight that these impacts are not only superficial but were detected down to 50-cm soil depth.

Decreased SOC, total N and extractable P in roads and logging decks were primarily caused by the removal of organic matter during logging (e.g., via decomposition or displacement) and absence of plant material inputs after logging. A similar cause was attributed by an earlier study on tropical deforestation, showing that the input of organic materials drives the dynamics and levels of C and N in soils (Sohng et al., 2017). Intense decomposition without fresh litter inputs can also explain the increased soil ^{15}N natural abundance in the roads and logging decks (Table 2.2), as implied by increases in ^{15}N signatures with soil depths (Baldos et al., 2015; Corre et al., 2010; Matson et al., 2017; Sotta et al., 2008). Other possible explanations can be soil erosion, overland runoff and leaching from roads and logging decks as cutting down the vegetation and compaction of the topsoil makes these sandy loam soils vulnerable to these processes of nutrient losses. Bulldozing of roads and logging decks could have also transported SOC, N and P to the sides, although our sampling design have included the entire breadth of these strata and there were no more visible piled soils at the sides by the time of our soil sampling. Additionally, such possible topsoil

displacement can even accelerate SOC decomposition through aggregate disruption (de Blécourt et al., 2014).

In the CL site, decreased exchangeable Al with increasing degree of disturbance (i.e., roads and logging decks) was due to the increase in soil pH relative to the reference area, which may be due to the release of base cations (Table 2.2) from decomposed organic matter and plant litter. With the increased pH, exchangeable Al may have been precipitated as Al hydroxide. Such decrease in exchangeable Al is therefore not a loss from this 50-cm depth but rather a chemical conversion to non-exchangeable form. Increase in exchangeable Mn in increased degree of disturbance may have been a consequence of a relatively high soil bulk density (Table 2.2). High soil bulk density may have promoted occurrence of reduced conditions, e.g., during high rainfall events, and in turn reduction of MnO_2 to the exchangeable form Mn^{2+} (Schlesinger and Bernhardt, 2013). Similar observations should been expected for exchangeable Fe, but at this low pH of our Ferralsol soil (Table 2.2) Fe can precipitate as Fe hydroxide while Mn still remains in exchangeable ionic form (Schlesinger and Bernhardt, 2013; page 258).

We expected increases in nutrient stocks in the felling gaps consequently from inputs of plant litter and woody debris of felled tree crown, but we did not detect significant change in nutrient stocks as compared to the reference areas (Table 2.2). This suggests that nutrient inputs from felled tree crown were not large enough to be detected against the initial stocks in the soil. A similar result was found in Amazonia, where the canopy foliage inputs represented only 0.1–0.7% of C and macro-nutrients compared to their original amount in the top 10-cm soil depth (Olander et al., 2005).

Although the decreases in area-weighted element stocks at the disturbed strata relative to the reference areas were substantial (decreases of 21–29%; Fig. 3b), these disturbed strata

only covered 4–5% of the logged forest sites (Table S2.1), largely because of the low logging intensity at our sites (Fig. S2.1). As opposed to the findings of Olander et al. (2005), where nutrient export from timber harvest was large, the low logging intensity at our sites also resulted in much lower nutrient export from timber removal (Table 2.3) compared to the soil nutrient losses (Fig. 2.3a). However, the promotion of new merchantable species of the Cameroonian timber market will increase logging intensity, and consequently nutrient export with timber harvest. This will also open more roads and logging decks (Fig. 2.2), where we observed decreases in SOC and nutrient element stocks (Table 2.2). This is likely to happen in CL system, as its lack of requirement for prior planning of harvesting operations (roads, logging decks and skidding trails) can increase mechanical disturbance of the forest area with increase in logging intensity (Fig. S2.2). Lastly, premature re-logging inherent to CL system can be detrimental to soils, as this can result in considerable increase in SOC loss after a second logging event (Chiti et al. 2016) but if avoided can save up to 34% of the forest carbon stocks (Sasaki et al., 2016).

2.6.   Conclusions

Selective logging affected soil characteristics down to 50-cm depth and the changes in SOC and nutrient stocks followed the spatial pattern of disturbance, with the highest changes in roads and logging decks, supporting our first hypothesis. At a very low logging intensity, the proportion of area disturbed by selective logging was very low in both the CL and RIL sites. After the first logging event, changes in the soil nutrient stocks were similar in the CL and RIL sites, rejecting our second hypothesis. Our results suggest that opening less roads than skidding trails to extract timber from the forest can reduce losses of SOC and nutrients, as lower changes in soil characteristics occurred in skidding trails. Minimizing the length of

roads will also result in less logging decks that were the second most impacted stratum after roads. However, as it is inherent for a CL system to re-log a forest frequently as market demands open-up, short logging rotations may exacerbate SOC and nutrient losses from the soil and nutrient export with timber harvest. Additionally, CL may influence climate change mitigation as its inherent short logging rotation could diminish carbon sink via net primary production. Our study suggests that RIL, which requires a management plan that prevents premature re-logging (e.g., a 30-year logging cycle at our RIL site), can be more sustainable than CL by allowing time for soil nutrients and vegetation to recover. Further research on cumulative effect of consecutive logging on soil characteristics and their recovery with time are required to improve knowledge on soil degradation following low-intensity logging.

2.7. Acknowledgments

2.8.   References

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2.9.   Appendix

Table S2.1

Area coverage [†] (and density [‡]) of the disturbed strata within the conventional selective logging and reduced-impact logging.

Logging system	Road	Logging deck	Skidding trail	Felling gap	Total	
	% (m ha^{-1})	% (# ha^{-1})	% (m ha^{-1})	% (# ha^{-1})	%	m^2 ha^{-1}
Conventional	1.44 (21.2)	0.24 (0.04)	2.42 (53.3)	1.07 (0.23)	5.17	517
Reduced-impact	1.19 (14.5)	0.10 (0.01)	1.76 (40.2)	0.98 (0.28)	4.03	403

[†] Ground disturbance expressed as percentage of the total logged forest area.

[‡] Number (#) or length (m) per unit of area.

Table S2.2

Element concentration (kg $m^{-3}_{wood\ fresh\ volume}$; mean $\pm$ SE, $n = 3$ individuals) in fresh wood of 16 timber species harvested in a rainforest of southern Cameroon.

Species	C	N	P	K	Mg	Ca
Afzelia bipindensis	367.97 ± 4.79	2.63 ± 0.14	0.07 ± 0.00	0.99 ± 0.06	0.28 ± 0.04	2.63 ± 0.15
Afzelia pachyloba	321.36 ± 34.86	1.67 ± 0.21	0.10 ± 0.02	0.72 ± 0.10	0.32 ± 0.07	0.99 ± 0.04
Dialium bipendensis	422.41 ± 1.40	1.81 ± 0.04	0.06 ± 0.01	0.60 ± 0.08	0.37 ± 0.00	1.45 ± 0.06
Entandrophragma utile	310.55 ± 2.80	1.41 ± 0.05	0.08 ± 0.01	0.70 ± 0.01	0.17 ± 0.01	0.99 ± 0.01
Erythropleum ivorense	467.02 ± 12.50	2.92 ± 0.15	0.04 ± 0.01	0.24 ± 0.04	0.05 ± 0.02	0.65 ± 0.06
Guibourtia ehie	354.08 ± 18.89	1.68 ± 0.03	0.03 ± 0.01	0.47 ± 0.07	0.02 ± 0.01	2.47 ± 0.46
Khaya grandifoliola	273.19 ± 5.17	0.84 ± 0.01	0.05 ± 0.00	0.50 ± 0.03	0.21 ± 0.01	0.95 ± 0.04
Klainedoxa gabonensis	340.80 ± 3.45	1.54 ± 0.07	0.07 ± 0.01	0.65 ± 0.14	0.67 ± 0.05	1.18 ± 0.10
Lophira alata	508.93 ± 6.72	2.02 ± 0.20	0.08 ± 0.04	0.51 ± 0.09	0.26 ± 0.04	2.47 ± 0.47
Lovoa trichilioides	309.86 ± 9.64	3.20 ± 0.31	0.03 ± 0.00	0.39 ± 0.06	0.11 ± 0.02	2.23 ± 0.27
Mitragyna ciliata	342.30 ± 9.78	1.59 ± 0.09	0.03 ± 0.01	1.68 ± 0.44	0.72 ± 0.03	3.50 ± 0.08
Nauclea diderrichii	381.41 ± 7.40	3.47 ± 0.16	0.04 ± 0.01	0.35 ± 0.02	0.08 ± 0.02	0.85 ± 0.09
Piptadeniastrum africanum	401.40 ± 7.55	5.56 ± 0.17	0.05 ± 0.01	0.64 ± 0.06	0.09 ± 0.01	1.65 ± 0.15
Pterocarpus soyauxii	372.35 ± 7.18	2.83 ± 0.27	0.05 ± 0.02	0.55 ± 0.05	0.14 ± 0.02	1.80 ± 0.41
Pycnanthus angolensis	303.39 ± 1.82	3.45 ± 0.08	0.03 ± 0.00	0.34 ± 0.03	0.06 ± 0.00	0.85 ± 0.01
Tetraberlinia bifoliolata	381.25 ± 15.33	2.33 ± 0.20	0.05 ± 0.01	0.44 ± 0.06	0.33 ± 0.08	1.81 ± 0.30

Element concentration (kg $m^{-3}_{wood\ fresh\ volume}$) = element concentration in kg $kg^{-1}_{dry\ mass}$ $\times$ wood specific gravity in $Mg_{dry\ mass}$ $m^{-3}_{fresh\ volume}$ $\times$ 1000, weighted by stem compartments
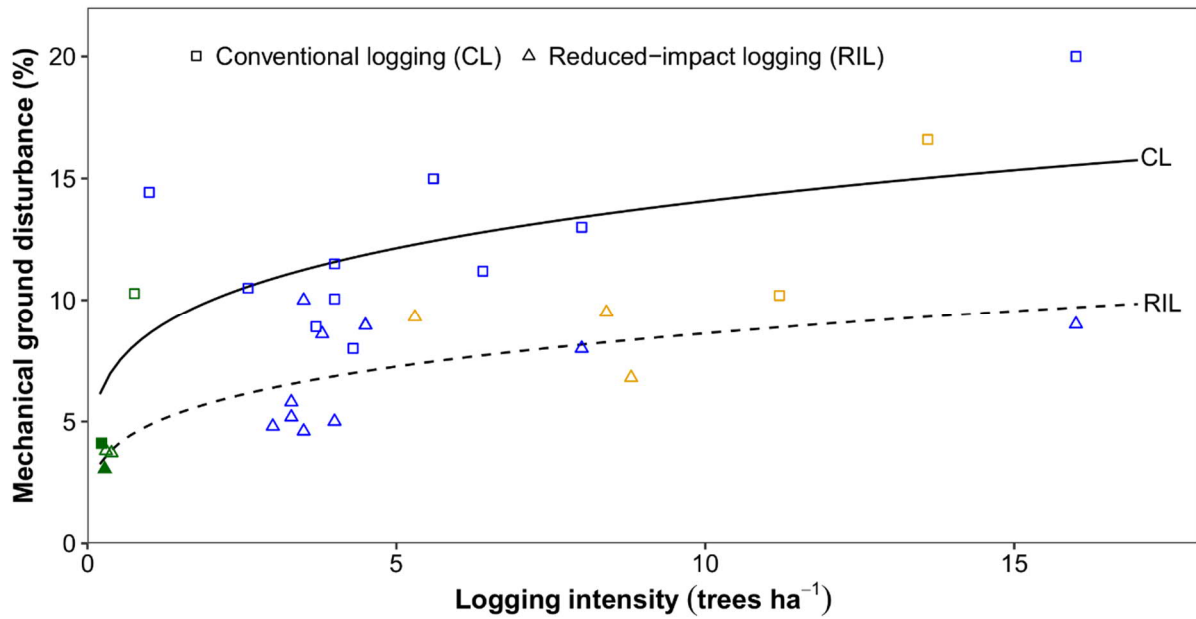
Fig. S2.1. Relationship between mechanical ground disturbance (roads + logging decks + skidding trails) and logging intensity in conventional selective logging and reduced-impact logging. Filled symbols are data from our present study; all the other data are from previous studies in Congo Basin (green), Amazonia (blue) and Malaysia (orange) (Asner et al., 2004; Holmes et al., 2002; Johns et al., 1996; Jonkers and van Leersum, 2000; Keller et al., 2005; Medjibe et al., 2013; Pereira et al., 2002; Pinard et al., 2000; Uhl and Vieira, 1989; van der Hout, 2000; Veríssimo et al., 1995).

CHAPTER 3

SOIL GREENHOUSE GAS FLUXES FOLLOWING CONVENTIONAL SELECTIVE LOGGING AND REDUCED-IMPACT LOGGING IN A CONGO BASIN RAINFOREST OF CAMEROON

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3.1. Abstract

Selective logging is among the main causes of tropical forest degradation, but little is known about its effects on greenhouse gas (GHG) fluxes from highly weathered soils in Africa. We measured soil CO_2, N_2O, and CH_4 fluxes (using vented static chambers), and their controlling factors in two forests that had undergone conventional selective logging and reduced-impact logging, in southern Cameroon. Each logging system had four plots and each plot included road, logging deck, skidding trail, felling gap, and an undisturbed reference area, wherein measurements were conducted monthly from September 2016 to October 2017. Annual GHG fluxes ranged from 4.9 to 18.6 Mg CO_2-C, 1.5 to 79 kg N_2O-N, and –4.3 to 71.1 kg CH_4-C ha^{-1} yr^{-1}. Soil CO_2 emissions were reduced and CH_4 was largely emitted from areas of soil compaction (skidding trails, logging decks and roads) ($P < 0.01$), whereas soil N_2O emissions increased only from the skidding trails ($P = 0.03–0.05$). The disturbed area showed overall decrease of 28% and increase of 83% in annual soil CO_2 and N_2O emissions respectively, with CH_4 emissions seven times higher than the uptake in the undisturbed area ($P \leq 0.01$). Across all strata, soil GHG fluxes were regulated by soil bulk density or WFPS that control soil aeration and gas diffusion, followed by soil organic matter that determine substrate availability for soil microbial processes. Our study suggests that the spatial pattern of logging disturbances is worthy to be considered when assessing selective logging impact on forest soils.

Keywords: *tropical forest, selective logging, soil CO_2 emissions, soil N_2O emissions, soil CH_4 fluxes, Ferralsols.*

3.2.   Introduction

Selective logging is the most common management practice used for timber harvesting in the tropics. In Cameroon, conventional selective logging and reduced-impact logging are commonly used. Conventional logging (CL) is generally carried without a requirement for prior planning of the field operations. In contrast, reduced-impact logging (RIL) has a management plan with a set measures (e.g., pre-logging forest inventory, mapping of merchantable trees, planning of logging infrastructures, directional tree-felling techniques, and post-harvest closure of roads and skidding trails) to minimize logging negative impacts (Putz et al., 2008). Both selective logging systems, however, can result in significant soil organic carbon (SOC) losses and decrease in soil fertility (Tchiofo Lontsi et al., 2019) which may lead to forest degradation. Carbon emissions associated with tropical forest degradation are estimated to be 2 Gt CO_2eq yr^{-1}, with selective logging being one of the largest contributor (Hosonuma et al., 2012; Pearson et al., 2017). This estimate took into account only the C removed from the forest in the form of harvested timber and harvest residues left on the forest floor to decompose. However, heavy equipment used to harvest timber also negatively impacts forest soils (e.g., soil compaction and removal of the organic-matter-rich topsoil), which can influence soil nutrient levels, aeration, and water infiltration and drainage (Hartmann et al., 2014; Olander et al., 2005; Schnurr-Pütz et al., 2006; Tchiofo Lontsi et al., 2019). These can, in turn, affect soil microbial communities and their function on greenhouse gas (GHG) regulation (Hartmann et al., 2014; Schnurr-Pütz et al., 2006).

The impact of selective logging in tropical forests depicts a gradient of disturbance from canopy opening and pulse litter input on felling gaps, soil compaction in skidding trails, logging decks, and roads, to vegetation and topsoil removal in logging decks and roads (Keller et al., 2005; Olander et al., 2005; Tchiofo Lontsi et al., 2019). Changes in soil physical and

biochemical properties follow this gradient, with larger changes in roads and logging decks and intermediate in skidding trails and felling gaps as compared to the undisturbed reference area (Olander et al., 2005; Tchiofo Lontsi et al., 2019). The proportion of forest area disturbed by selective logging is related to the tree harvest intensity (Pereira Jr. et al., 2002; Tchiofo Lontsi et al., 2019). Where only few trees are harvested per hectare (e.g., in Africa), a large proportion of the logged forest can remain undisturbed (Putz et al., 2019; Tchiofo Lontsi et al., 2019). This undisturbed forest area may remain functionally intact as far as biogeochemical processes are concerned. For example, a study carried out in a Brazilian rainforest found no significant difference in soil GHG fluxes between the undisturbed areas within the selectively logged forest and the nearby unlogged forest (Keller et al., 2005).

Soil physico-chemical properties, together with climatic factors and forest management systems, are among the key spatial and temporal drivers of soil GHG emissions from natural tropical forest ecosystems (Kim et al., 2016; Koehler et al., 2009a, 2009b; Matson et al., 2017). Soil CO_2 emissions from some African tropical ecosystems showed positive relationships with both soil moisture and temperature (Kim et al., 2016; MacCarthy et al., 2018; Wanyama et al., 2019; Werner et al., 2007). Additionally, soil characteristics like bulk density (Goutal et al., 2012; Liu et al., 2014; Wanyama et al., 2019), SOC, total N, and extractable P (Hassler et al., 2015; Liu et al., 2014; Schwendenmann et al., 2003; Werner et al., 2007) explain spatial variations in soil CO_2 emissions. Forest logging disturbances from harvest machinery that reduce roots and litter inputs, and decreases SOC and aeration (e.g., compaction or increase in soil bulk density) can reduce soil CO_2 emissions (Han et al., 2015; Mori et al., 2017; Zerva and Mencuccini, 2005).

Moreover, soil N_2O emissions from tropical forest soils increase with increases in soil N availability and moisture contents (Corre et al., 2014; Davidson et al., 2000a, 2004; Hassler et al., 2017; Keller et al., 2005; Koehler et al., 2009b; Matson et al., 2017; van Lent et al.,

2015; Werner et al., 2007). Forest logging activities that alter soil N availability, microclimate, and aeration (compaction or increase in soil bulk density) can change soil N_2O fluxes (Keller et al., 2005; Yashiro et al., 2008). For example, more compacted soils of logging deck and skidding trail emitted higher N_2O than undisturbed forest (Keller et al., 2005). Another study, although it did not consider the logging damage strata in the experimental design also found larger soil N_2O emissions from the selectively logged forest compared to unlogged forest, as a result of increased soil mineral N content and soil compaction (Yashiro et al., 2008).

CH_4 flux from the soil surface is a net result from methanogenesis and methanotrophy, which can occur concurrently in soils (Hassler et al., 2015; Matson et al., 2017; Veldkamp et al., 2013). Across an orthogonal gradient of precipitation and soil fertility, highly weathered Ferralsol soils with intermediate annual rainfall (2360-2690 mm yr^{-1}) but high soil N availability have larger CH_4 uptake, followed by Ferralsols with high rainfall (3400 mm yr^{-1}), than Cambisol soils with low annual rainfall (1700-2030 mm yr^{-1}) and low N availability (Matson et al., 2017). This showed that soil CH_4 flux is largely controlled by soil N availability that influences methanotrophic activity (Bodelier and Laanbroek, 2004), as well as by precipitation or soil moisture, through its influence on gas diffusivity (for methanotrophs) and anaerobicity (for methanogens) (Davidson et al., 2004; Matson et al., 2017; Veldkamp et al., 2013; Verchot et al., 2000; Werner et al., 2007). Thus, forest disturbance like logging, which changes soil fertility and increase soil bulk density (Tchiofo Lontsi et al., 2019), can change soil CH_4 fluxes. For example, conversion of tropical lowland forests to unfertilized agricultural systems decreases soil N availability which, in turn, decreases CH_4 uptake in the soil (Gütlein et al., 2018; Hassler et al., 2015). Moreover, soil compaction from intensive agriculture and silviculture decreases CH_4 uptake (Wanyama et al., 2019). Also, logged forest soils tend to emit CH_4 especially from disturbed areas with strong soil compaction whereas

undisturbed well-drained forest soils consume methane (Keller et al., 2005; Yashiro et al., 2008; Zerva and Mencuccini, 2005).

Studies on soil GHG fluxes in Africa were mostly done in agricultural lands (e.g., MacCarthy et al., 2018; Pelster et al., 2017; Rosenstock et al., 2016; van Straaten et al., 2019; Wanyama et al., 2019). Only a handful of studies have investigated soil GHG fluxes from tropical rainforests in Africa (Castaldi et al., 2013; Gütlein et al., 2018; Werner et al., 2007). A few studies estimated C emissions from selective logging in African tropical forests, but they focused mainly on vegetation biomass-C losses and did not include C emissions from the soil (e.g., Pearson et al., 2017; Umunay et al., 2019). Recognizing this knowledge gap, we conducted this study in order to find out the spatial changes in soil GHG fluxes following selective logging in an African forest on highly weathered Ferralsol soil that developed on pre-Cambrian basement rocks. We conducted monthly measurements of soil GHG fluxes and their controlling factors for one year in two lowland rainforests that had undergone conventional selective logging and reduced-impact logging. Our objectives were to (1) assess the changes in soil CO_2, N_2O and CH_4 fluxes resulting from these selective logging systems, and (2) determine the spatial and temporal controlling factors of the changes in soil GHG fluxes in these selectively logged forests. We included in the experimental design the spatial pattern of logging disturbances (felling gaps, skidding trails, logging decks, and roads) and compared them to undisturbed area within the forests. We hypothesized that (1) soil CO_2 emissions will decrease while soil N_2O emissions will increase from undisturbed reference area to highly disturbed logging decks and roads, and soil CH_4 fluxes will turn from net consumption in undisturbed area to net emissions from these highly disturbed areas. (2) Along the gradient of soil disturbance from felling gaps to roads, decreases in SOC and N availability (due to vegetation and topsoil removal and decomposition of left-over litter) and increases in

soil compaction (which influences soil water content or aeration and gas diffusion) will determine the spatial and temporal patterns of these soil GHG fluxes.

3.3. Material and methods

3.3.1. Study sites and experimental design

Soil greenhouse gas fluxes were measured in a lowland rainforest (30-100 m above sea level) located in the Technical Operational Unit (TOU) Campo-Ma'an, in the southern Cameroon (2°10'–2°52'N, 9°50'–10°54'E). The geological parent material consists of metamorphic micaschists, superior and inferior gneiss and undifferentiated gneiss formed on inferior pre-Cambrian basement rocks (Gwanfogbe et al., 1983). The soils are heavily weathered, sandy loam Ferralsols with acidic pH, low effective cation exchange capacity (ECEC) and base saturation, and high aluminum (Al) saturation (Table S3.1; Tchiofo Lontsi et al., 2019). The climate is equatorial with a bi-modal rainfall pattern defining two dry seasons, usually from December to February and July to August, when monthly rainfall is less than 150 mm. The mean annual rainfall is 2693 mm with a mean annual air temperature of 25.4°C (Climate-Data.org, 2018). During our study period (from September 2016 to October 2017), the soil temperature ranged from 23.9 to 30.7°C. The vegetation in our study sites is dominated by *Sacoglottis gabonensis* and the most common harvested timber species were *Lophira alata*, *Erythropleum ivorense*, *Guibourtia ehie*, and *Pterocarpus soyauxii* (Tchiofo Lontsi et al., 2019).

We selected two sites with a relatively flat topography, located approximately 40 km apart that were selectively logged six to seven months prior to the start of our study. The first site was located in the multipurpose area of the TOU Campo-Ma'an (Fig. 1.1), in Mintom village, which is approximately 4 km east of the city of Campo. This 750-ha forest site was

logged under a local agreement between the village elders and a private operator, so as no management plan was required. Using so-called conventional selective logging (CL) practices, the logger located the desired trees that were harvested with minimal planning and little consideration to the remaining stand. The logging intensity in the CL site was 2.75 m^3 ha^{-1}, which resulted in an area disturbed from logging (the sum of felling gap, skidding trail, logging deck and road) of 5.2% of the total forest area (Tchiofo Lontsi et al., 2019). The second site (2350 ha) was located in a logging concession (Fig. 1.1) owned by a commercial enterprise that holds a Forest Stewardship Council (FSC) certification of sustainable forest management. Prior to logging, a forest management plan with a 30-year logging cycle was required and logging operations fulfilled the FSC certification standards. The logging company used reduced-impact selective logging (RIL) protocols that consisted of pre-harvest tree inventory, planning of skidding trail and road locations, controlled tree felling, and the employment of trained and qualified field crews. The logging intensity in the RIL site was 2.78 m^3 ha^{-1} and the coverage of the disturbed area from logging was 4% of the total forest area (Tchiofo Lontsi et al., 2019).

In each logging system (CL or RIL), we conducted our measurements in four disturbed strata, i.e., felling gap, skidding trail, logging deck and road, and in an undisturbed reference area. This reference area was selected at least 50 m away from any disturbed areas. Roads and logging decks are the most affected by logging operations, where ECEC, SOC, total N, and Bray-extractable P decrease whereas pH and ^{15}N natural abundance increase compared to the undisturbed reference area (Table S3.1; Tchiofo Lontsi et al., 2019). At the CL and RIL sites, we established four replicate plots centered around four randomly selected logging decks, and each plot encompassed the five above-mentionned strata (Fig. S3.1). The distance between replicate plots at each site was at least 500 m.

3.3.2. Soil greenhouse gas fluxes

Soil CO_2, N_2O, and CH_4 fluxes were measured monthly from September 2016 to October 2017, using vented static chambers (e.g., Hassler et al., 2015; Koehler et al., 2009a; Matson et al., 2017; van Straaten et al., 2019). In order to represent the range of conditions in each stratum, we installed four chamber bases (0.04 m^2 area, 0.25 m total height, inserted into the soil at approx. 0.02 m depth, and 11 L total volume with cover) for each stratum of each replicate plot. The chamber bases were closed with polyethylene covers, equipped with Luer-lock sampling ports on the center top. From each chamber, four gas samples were taken using syringes over a 30-minute sampling period (at 2, 12, 22, and 32 minutes following chamber closure). Gas samples were stored with overpressure into pre-evacuated 12 mL glass vials (Labco Exetainers, Labco Limited, Lampeter, UK) with rubber septa. On each measurement month, a total of 640 gas samples (4 time intervals × 4 chambers × 5 strata × 4 replicate plots × 2 logging systems) were taken and transported by air every 4–5 months to the laboratory at the University of Goettingen, Germany. These exetainers have been proven in a number of studies conducted by our group to be leak proof (e.g., Hassler et al., 2015; Matson et al., 2017; van Straaten et al., 2019).

The gas samples were analyzed using a gas chromatograph (GC; SRI 8610C, SRI Instruments, Torrance, CA, USA) equipped with a flame ionization detector to measure CH_4 concentrations and CO_2 (with methanizer) as well as an electron capture detector for N_2O measurement with make-up gas of 5% CO_2 – 95% N_2. Three calibration gases (Deuste Steininger GmbH, Mühlhausen, Germany) were used to calibrate the GC prior to each analysis with concentrations ranging from 400 to 3000 ppm for CO_2, 360 to 1600 ppb for N_2O and 1000 to 5000 ppb for CH_4. Soil GHG fluxes were calculated from the linear change in concentration with chamber closure time, and adjusted with the field-measured air

temperature and atmospheric pressure during sampling (Koehler et al., 2009b). For soil N_2O and CH_4 fluxes, all zero and negative fluxes were included in our data analysis.

Annual soil CO_2, N_2O and CH_4 fluxes were estimated based on trapezoidal rule between measured fluxes and sampling day intervals, assuming constant flux rates per day (Hassler et al., 2015; Koehler et al., 2009a, 2009b; Matson et al., 2017; Veldkamp et al., 2013). For each replicate plot, we calculated the overall annual CO_2, N_2O or CH_4 fluxes from the four disturbed strata by weighting them with their percentage areal coverage (i.e., felling gap: 1.0–1.1%, skidding trail: 1.7–2.4%, logging deck: 0.1–0.2%, road: 1.2–1.4% in RIL and CL; Tchiofo Lontsi et al., 2019). These values were then subtracted by the annual CO_2, N_2O or CH_4 fluxes from the undisturbed reference area in each replicate plot to get an overall change of these GHG fluxes as a consequence of logging, similar to the method of Keller et al. (2005).

3.3.3. Soil controlling factors

Following each soil GHG flux measurement, soil temperature, moisture, and mineral N concentration were measured in the top 5 cm mineral soil. We were able to measure mineral N only during the last six months of the fieldwork, as the needed chemicals shipped from Germany arrived late due to administrative issues in Cameroon, involving custom clearance of shipped supplies. Soil temperature was measured near each chamber base using a portable thermometer with probe (Greisinger GMH 3210, Greisinger Messtechnik GmbH, Regenstauf, Germany).

Soil in the top 5-cm depth was sampled at about 1-m away from each of the four chamber bases per stratum for mineral N extraction and soil moisture determination. The four soil sub-samples were then thoroughly mixed to have one composite sample for each specific

stratum in each replicate plot. One part of these soil samples was used for soil mineral N extraction, which was done in-situ to avoid changes in mineral N concentrations due to storage of disturbed soil samples (Arnold et al., 2008). In the field, freshly sampled soils were added into prepared extraction bottles (250 mL plastic bottles containing 150 mL of 0.5 M K_2SO_4 solution) and shaken thoroughly. Upon arrival at the local field station, the bottles were shaken again for one hour and filtered. The filtered extracts were stored in 20 mL scintillation vials and immediately frozen for transport to the University of Goettingen, Germany. In the laboratory of Goettingen University, soil extracts were analyzed for total extractable N, NH_4^+, and NO_3^- using continuous flow injection colorimetry (SEAL Analytical AA3). Total extractable N was determined by ultraviolet-persulfate digestion followed by hydrazine sulfate reduction (Autoanalyzer Method G-157-96), NH_4^+ by salicylate and dicloro isocyanuric acid reaction (Autoanalyzer Method G-102-93), and NO_3^- by cadmium reduction method with NH_4Cl buffer (Autoanalyzer Method G-254-02). Soil moisture content was determined from the remaining soil samples upon arrival at the field station in Cameroon. Gravimetric moisture content was measured by oven-drying soils at 105°C for 24 h and was expressed as water-filled pore space (WFPS) using the measured bulk density (Table S3.1) and the particle density of mineral soil (2.65 g cm^{-3}). The gravimetric moisture content was also used to calculate the dry mass of soil extracted for mineral N.

Soil physical and biochemical characteristics in the top 50-cm depth were reported in our previous study (Tchiofo Lontsi et al., 2019); and those in the top 10-cm depth are reports in Table S1. Soil sampling and analysis are described in detail in the supporting information (Text S3.1).

3.3.4. Statistics

Statistical tests of the repeatedly measured soil GHG fluxes and soil parameters (WFPS, soil temperature, and mineral N concentrations) were conducted using linear mixed-effects (LME) models followed by Fisher's least significant difference (LSD) test at $P \leq 0.05$. These tests were carried out on the average of the four chambers (as subsamples) on a given sampling day, representing each stratum within each replicate plot, and conducted across all sampling days. Each parameter was first checked for normality (Shapiro-Wilk test) and in cases of non-normal distribution, we used a logarithmic (e.g., CH_4, soil temperature, mineral N) or a square root (e.g., CO_2, NO_3^-) transformation. For LME tests, either strata (when comparing among road, logging deck, skidding trail, felling gap and reference area for each logging system) or logging systems (when comparing between CL and RIL) were used as the fixed factor whereas replicate plots and sampling days were included as random factors. If the relative goodness of the model (based on the Akaike information criterion) was improved, we included in the model heteroscedasticity of the fixed-factor variances (using varIdent function) and/or the decreasing autocorrelation of sampling days with increasing time interval (using corAR1 function). The LME model was also used to assess seasonal difference in soil GHG fluxes and soil parameters for each stratum (first, separately for each logging system, and then combined when there were no difference between logging systems) with season as the fixed factor and the random factors as above.

To assess the relationships with temporal and spatial soil controlling factors of soil GHG fluxes, we conducted Spearman's rank correlation tests. We conducted the correlation tests across logging systems, as almost all soil variables that correlated with the soil GHG fluxes did not differ between CL and RIL. To assess how soil factors (WFPS, soil temperature, and mineral N) influenced the temporal variations of soil GHG fluxes, we used the means of the four replicate plots for each stratum on each sampling day and tested their correlations

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across the 12-month measurement period. Correlations were assessed for the undisturbed reference area ($n = 24$ (2 logging systems $\times$ 12 months), except for mineral N with six monthly measurements for which $n = 12$), and across all strata ($n = 120$ (2 logging systems $\times$ 5 strata $\times$ 12 months), except for mineral N for which $n = 60$). To determine the influence of soil physical and biochemical characteristics (Table S3.1) on the spatial variations of soil GHG fluxes, we used the annual GHG flux for each replicate plot of the undisturbed reference area ($n = 8$ (2 logging systems $\times$ 4 plots)) and across all strata ($n = 40$ (2 logging systems $\times$ 4 plots 5 strata)). Correlation coefficients were considered significant at $P \leq 0.05$. All statistical analyses were performed using the open-source statistical software R version 3.5.1 (R Core Team, 2018).

3.4. Results

3.4.1. Soil GHG fluxes and controlling factors in the undisturbed reference areas

The undisturbed reference areas of CL and RIL consistently emitted N_2O and consumed CH_4 throughout the measurement period (Fig. 3.1c–f). Across the 12-month measurement period, soil CO_2, N_2O and CH_4 fluxes from the undisturbed reference areas did not differ between CL and RIL ($P = 0.15$–0.41; Fig. 3.1a–f; Table 3.1). We also did not find any difference in the three GHG fluxes between dry and wet seasons in each or both logging systems ($P = 0.13$–0.20). Soil CO_2 emissions were 208 ± 6 and 181 ± 8 mg C m^{-2} h^{-1} in the wet and dry seasons, respectively; N_2O emissions in wet and dry seasons were 24 ± 2 and 13 ± 1 µg N m^{-2} h^{-1}, respectively; and CH_4 uptake was -34 ± 47 µg C m^{-2} h^{-1} in the wet season and -54 ± 6 µg C m^{-2} h^{-1} in the dry season.

Soil temperature, NH_4^+ and NO_3^- contents did not differ between CL and RIL undisturbed areas as well as between dry and wet seasons ($P = 0.17$–0.94; Fig. 3.2c and d;

Table 3.2). However, WFPS was larger in the wet than the dry season ($P < 0.01$; Fig. 3.2a and b; Table 3.2). Monthly soil CO_2 emissions from the undisturbed reference areas across CL and RIL sites were positively correlated (*Spearman's rho (r)* = 0.53, $P \leq 0.01$, $n = 24$) with monthly WFPS that ranged from 23–75% with an average of 45% (Fig. 3.2a and b; Table 3.2). Similarly, monthly WFPS were positively correlated with monthly soil N_2O emissions ($r = 0.53$, $P \leq 0.01$, $n = 24$) as well as monthly soil CH_4 fluxes ($r = 0.74$, $P \leq 0.01$, $n = 24$). Additionally, we found a negative correlation between soil CH_4 fluxes and NO_3^- ($r = -0.70$, $P = 0.01$, $n = 12$). Furthermore, annual soil CH_4 uptake was negatively correlated with bulk density ($r = -0.70$, $P = 0.05$, $n = 8$).
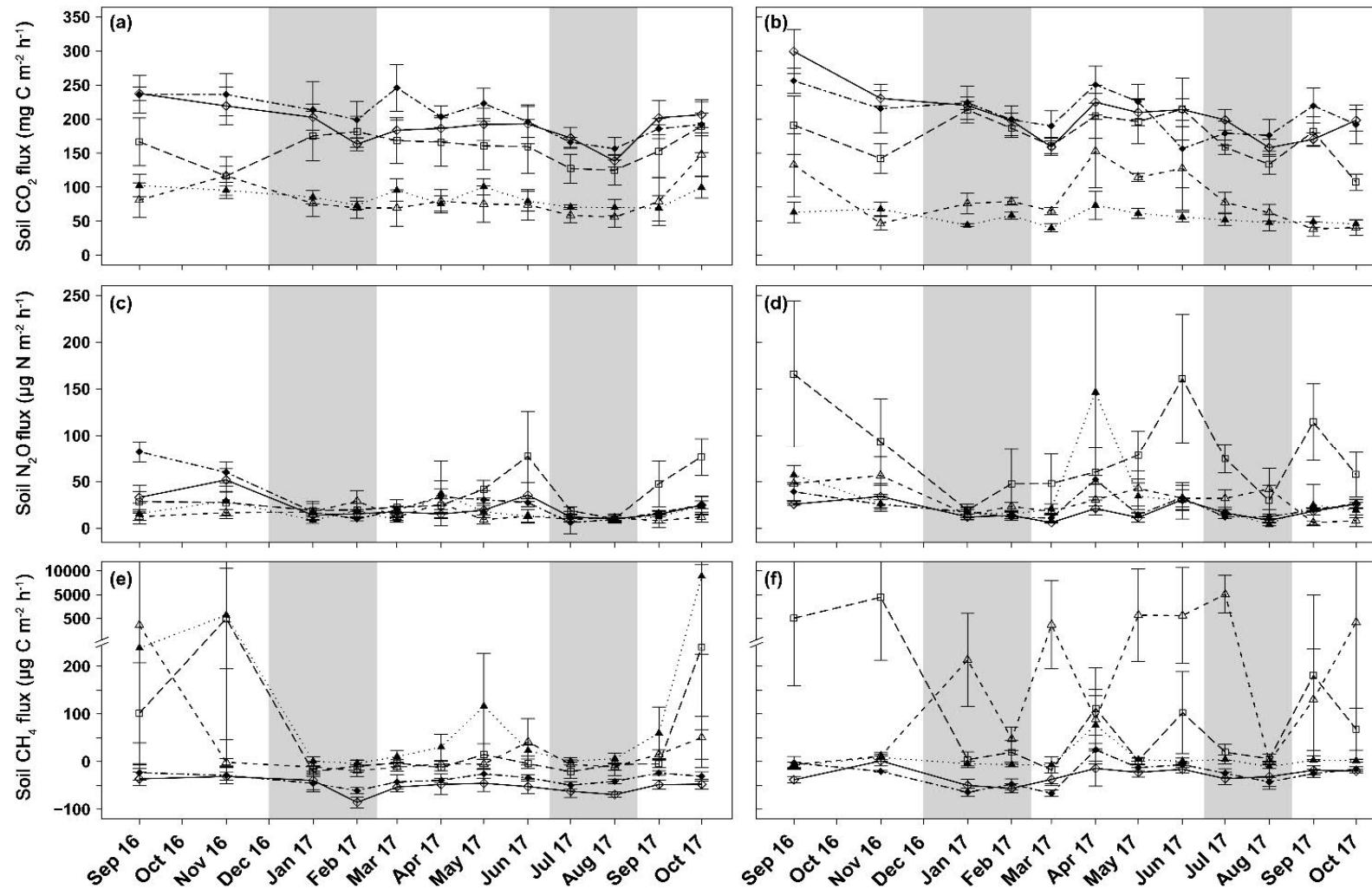
Fig. 3.1. Soil CO_2, N_2O and CH_4 fluxes (mean ± SE, $n = 4$ plots) from the undisturbed reference area (◊), felling gap (♦), skidding trail (□), logging deck (Δ), and road (▲) in conventional selective (a, c and e) and reduced-impact logging (b, d and f) in a Congo Basin rainforest of Cameroon. Grey shadings mark the dry season.
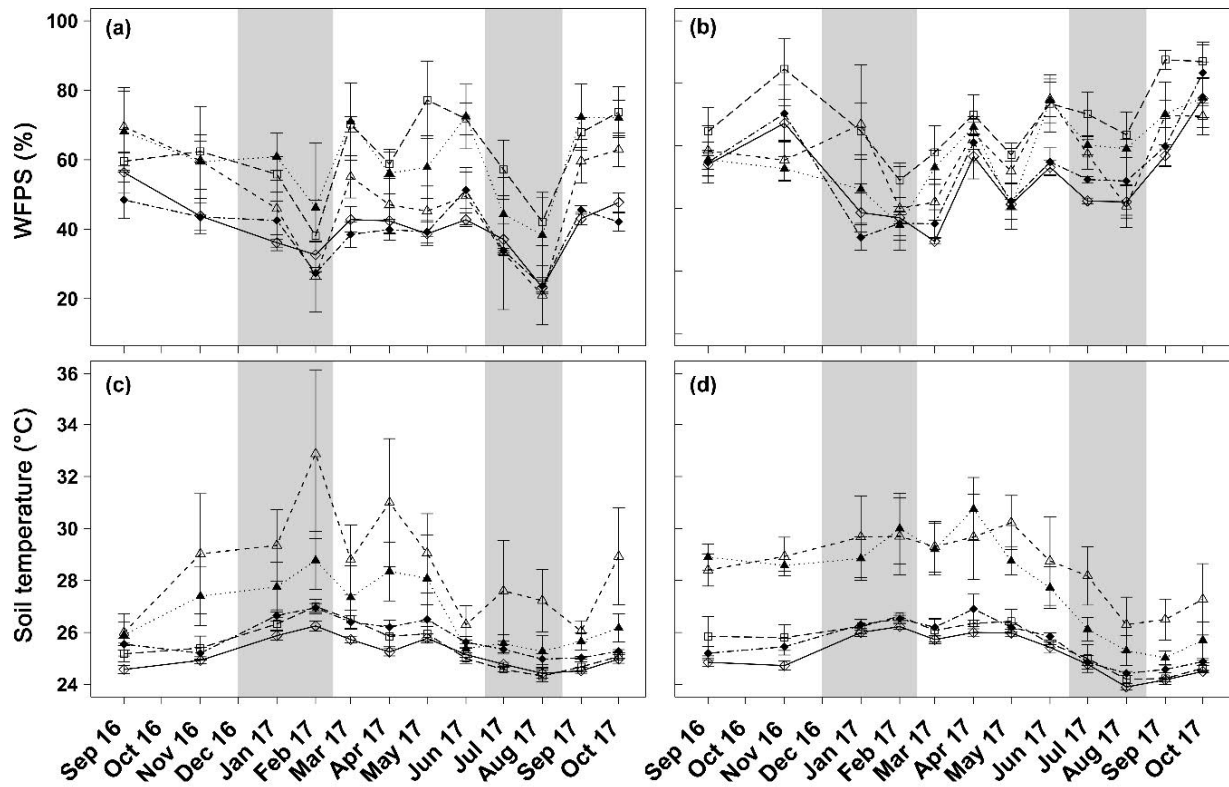
Table 3.1

Soil CO_2, N_2O and CH_4 fluxes from undisturbed reference area and disturbed strata following conventional selective and reduced-impact logging in a Congo Basin rainforest of Cameroon.

Logging system / Strata	CO_2 emission $(mg\ C\ m^{-2}\ h^{-1})$	Annual CO_2 emission $(Mg\ C\ ha^{-1}\ yr^{-1})$	N_2O emission $(\mu g\ N\ m^{-2}\ h^{-1})$	Annual N_2O emission $(kg\ N\ ha^{-1}\ yr^{-1})$	CH_4 fluxes $(\mu g\ C\ m^{-2}\ h^{-1})$	Annual CH_4 fluxes $(kg\ C\ ha^{-1}\ yr^{-1})$
Conventional logging						
Reference	191 ± 6^a	17.0 ± 1.0	22 ± 3^{ab}	2.1 ± 0.4	-52 ± 4^c	-4.3 ± 0.8
Felling gap	204 ± 8^a	18.2 ± 1.2	28 ± 4^{ab}	2.7 ± 0.3	-37 ± 3^c	-3.2 ± 0.8
Skidding trail	157 ± 9^b	13.5 ± 2.2	34 ± 6^a	2.7 ± 0.3	72 ± 39^{ab}	8.3 ± 4.1
Logging deck	82 ± 7^c	7.2 ± 1.4	15 ± 2^b	1.3 ± 0.3	42 ± 40^b	3.3 ± 3.5
Road	$85 \pm 4^{c,A}$	7.5 ± 0.7	17 ± 3^b	1.5 ± 0.5	873 ± 721^a	46.2 ± 44.1
Reduced-impact logging						
Reference	206 ± 7^a	18.3 ± 1.1	18 ± 2^b	1.6 ± 0.3	-29 ± 5^c	-2.5 ± 0.8
Felling gap	207 ± 8^a	18.6 ± 0.7	23 ± 2^b	2.0 ± 0.2	-26 ± 5^c	-2.4 ± 0.4
Skidding trail	174 ± 8^b	15.4 ± 1.0	79 ± 12^a	6.7 ± 2.3	469 ± 359^{ab}	71.1 ± 59.3
Logging deck	84 ± 9^c	7.2 ± 0.9	29 ± 4^b	2.8 ± 0.8	726 ± 339^a	51.1 ± 31.2
Road	$55 \pm 3^{d,B}$	4.9 ± 0.4	34 ± 10^b	2.9 ± 1.1	5 ± 6^b	0.5 ± 0.5

Measurements were conducted monthly from September 2016 to October 2017. Means ($\pm$ SE, $n = 4$ plots) followed by different lowercase letters indicate significant differences among strata within a logging system and different uppercase letters indicate significant differences between logging systems within a stratum (linear mixed-effect models with Fisher's LSD test at $P \leq 0.05$). Annual soil CO_2, N_2O and CH_4 fluxes were not statistically tested for differences between strata or logging systems since these annual values are trapezoidal extrapolations.

Fig. 3.2. Soil water-filled pore space (WFPS) and soil temperature (mean ± SE, *n* = 4 plots) in the top 5-cm depth of the undisturbed reference area (◊), felling gap (♦), skidding trail (□), logging deck (Δ), and road (▲) in conventional (a and c) and reduced-impact (b and d) selective logging in a Congo Basin rainforest of Cameroon. Grey shadings mark the dry season.

Table 3.2

Soil factors in dry and wet seasons for the undisturbed reference area and disturbed strata in both logging systems in a Congo Basin rainforest of Cameroon.

Seasons / Strata	WFPS (%)	Temperature (°C)	NH_4^+ (mg N kg^{-1})	NO_3^- (mg N kg^{-1})
Wet season				
Reference	$49.5 \pm 1.7^{c,A}$	25.1 ± 0.1^c	3.2 ± 0.3^a	0.6 ± 0.1^a
Felling gap	$50.6 \pm 2.0^{c,A}$	25.7 ± 0.1^c	3.9 ± 0.4^a	0.9 ± 0.2^a
Skidding trail	$70.2 \pm 1.9^{a,A}$	25.5 ± 0.1^c	2.8 ± 0.2^{ab}	0.5 ± 0.1^{ab}
Logging deck	$58.4 \pm 2.1^{b,A}$	28.4 ± 0.4^a	2.7 ± 0.7^{bc}	0.1 ± 0.0^b
Road	63.6 ± 2.0^a	27.4 ± 0.3^b	1.8 ± 0.2^c	0.1 ± 0.1^b
Dry season				
Reference	$36.1 \pm 1.5^{b,B}$	25.3 ± 0.2^c	3.1 ± 0.4^a	3.2 ± 1.2^a
Felling gap	$36.4 \pm 2.2^{b,B}$	25.8 ± 0.2^c	2.9 ± 0.4^a	2.2 ± 0.7^a
Skidding trail	$55.1 \pm 3.0^{a,B}$	25.5 ± 0.2^c	2.5 ± 0.3^{ab}	1.0 ± 0.4^{ab}
Logging deck	$41.4 \pm 4.6^{b,B}$	28.9 ± 0.6^a	2.4 ± 0.5^{ab}	0.3 ± 0.1^b
Road	48.7 ± 3.6^{ab}	27.2 ± 0.4^b	1.7 ± 0.2^b	0.2 ± 0.1^b

Soil factors were measured in the top 5-cm depth. Means ($\pm$SE, $n = 8$ plots) followed by different lowercase letters indicate significant differences among strata within each season, and different uppercase letters indicate significant differences between seasons within each stratum (linear mixed-effect models with Fisher's LSD test at $P \leq 0.05$).

3.4.2. Effect of selective logging on soil GHG fluxes and controlling factors

For all soil GHG fluxes, the patterns of changes in the disturbed strata as compared to the reference areas were similar in both CL and RIL ($P < 0.01$–0.05; Fig. 3.1a–f; Table 3.1). Soil GHG fluxes were largely altered in roads, logging decks and skidding trails, whereas felling gaps showed comparable fluxes with the undisturbed reference area. For roads, there were larger soil CO_2 emissions from the CL than RIL ($P < 0.01$; Table 3.1). Nonetheless, the overall changes in soil GHG fluxes from the disturbed strata (weighted by their respective areal

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coverages) relative to the undisturbed reference area did not differ between CL and RIL ($P = 0.06–0.92$).

Soil CO_2 emissions decreased in skidding trails, logging decks and roads as compared to the undisturbed reference area in CL and RIL ($P < 0.01$; Fig. 3.1a and b; Table 3.1), and were similar in the dry and wet seasons ($P = 0.08–0.64$). Overall annual soil CO_2 emissions from the disturbed strata (12.7 ± 0.6 Mg C ha^{-1} yr^{-1}) decreased by 28% in comparison to the undisturbed reference area (17.6 ± 1.1 Mg C ha^{-1} yr^{-1}) in both logging systems ($P < 0.01$; Table 3.1).

Soil N_2O emissions from the skidding trails were higher than the other disturbed strata ($P = 0.03–0.05$; Table 3.1). Seasonal pattern showed larger soil N_2O emissions in the wet season than in the dry season for felling gaps and skidding trails ($P < 0.01–0.05$; Fig. 3.1c and d). In both logging systems, annual soil N_2O emissions from the disturbed strata as a whole (3.3 ± 0.6 kg N ha^{-1} yr^{-1}) were 83% higher than the undisturbed reference area (1.8 ± 0.2 kg N ha^{-1} yr^{-1}; $P = 0.01$; Table 1).

Soil CH_4 fluxes were larger in the highly disturbed strata compared to the undisturbed reference area in CL and RIL ($P < 0.01$; Fig. 3.1e and f; Table 3.1). Despite high spatial and temporal variability (i.e., large standard error; Fig. 3.1e and f; Table 3.1), skidding trails and roads displayed net CH_4 uptake during the dry season (-1.1 ± 6.0 and -1.7 ± 2.4 µg C m^{-2} h^{-1}, respectively) and net emission during the wet season (406 ± 270 and 660 ± 541 µg C m^{-2} h^{-1}, respectively). The logging decks emitted CH_4 throughout, with 243 ± 90 and 666 ± 490 µg C m^{-2} h^{-1} during the dry and wet seasons, respectively. The overall annual soil CH_4 emissions from the disturbed area (24.1 ± 13.2 kg C ha^{-1} yr^{-1}) were seven times higher than the annual soil CH_4 consumption in the undisturbed reference area (-3.4 ± 0.6 kg C ha^{-1} yr^{-1}) in both logging systems ($P < 0.01$; Table 3.1).

The disturbed strata generally showed comparable WFPS between CL and RIL (P = 0.15–0.41; Fig. 3.2a and b). Among strata, WFPS was higher in skidding trails, roads, and logging decks compared to the undisturbed reference area, while felling gaps were comparable with the reference area ($P \leq 0.01$; Fig. 3.2a and b; Table 3.2). Seasonally, WFPS was higher in the wet than in dry season ($P < 0.01$–0.03; Table 3.2), except for the roads that showed no seasonal differences. Soil temperature was higher in logging decks and roads compared to skidding trails, felling gaps and the reference areas ($P < 0.01$; Fig. 3.2c and d; Table 3.2) and did not vary between seasons (P = 0.47–0.85; Table 3.2). Total extractable mineral N ($NH_4^+ + NO_3^-$) was lower in roads (2.0 ± 0.1 mg N kg^{-1}) and logging decks (2.7 ± 0.5 mg N kg^{-1}) compared to felling gaps (4.9 ± 0.4 mg N kg^{-1}) and the reference areas (4.6 ± 0.6 mg N kg^{-1}), whereas skidding trails showed intermediate values (3.4 ± 0.3 mg N kg^{-1}) ($P < 0.01$). Extractable mineral N did not show a seasonal variation and its dominant form was NH_4^+ in all the strata (Table 3.2).

3.4.3. Temporal and spatial controls of soil GHG fluxes across strata and logging systems

Instead of the positive relationship of soil CO_2 emissions with WFPS in the undisturbed area (see Section 3.4.1), there was a negative correlation with WFPS during the wet season with a maximum WFPS of 88% (r = –0.25, $P \leq 0.05$, n = 80). When we excluded the strata with highly compacted soil (i.e., increased soil bulk densities in logging decks and roads; Table S3.1), soil CO_2 emissions exhibited a parabolic relationship with WFPS in skidding trails, felling gaps and undisturbed area, with increasing soil CO_2 emissions from 23–44% WFPS (r = 0.39, P = 0.03, n = 31) and decreasing soil CO_2 emissions from 46–88% WFPS (r = –0.31, P = 0.05, n = 41). Moreover, we detected a negative correlation of soil CO_2 emissions

with soil temperature across the whole year, but by plotting the soil CO_2 emissions against soil temperature this showed that this negative correlation was caused by the reduced CO_2 emissions from logging decks and roads where soil temperatures had increased (Table 3.2) but SOC stocks had decreased (Table S3.1). Considering only the skidding trails, felling gaps and undisturbed area, soil CO_2 emissions were positively correlated with soil temperature across the whole-year measurements ($r = 0.27$, $P = 0.02$, $n = 72$), whereas there was no correlation observed when only considering logging decks and roads ($r = 0.21$, $P = 0.14$, $n = 48$). Soil CO_2 emissions showed positive correlations with the total extractable mineral N ($NH_4^+ + NO_3^-$) across the measurement period ($r = 0.46$, $P < 0.01$, $n = 60$). Furthermore, soil N_2O emissions were positively correlated with WFPS across the whole-year measurements ($r = 0.40$, $P < 0.01$, $n = 120$). Additionally, there was a positive correlation between soil N_2O emissions and NH_4^+ content when considering only the logging decks and roads ($r = 0.44$, $P = 0.03$, $n = 24$). Soil CH_4 fluxes were also positively correlated with WFPS across the measurement period ($r = 0.24$, $P = 0.01$, $n = 120$).

Annual soil CO_2 emissions were correlated positively with SOC, total N, Bray-extractable P, and ECEC and negatively with soil bulk density (Table 3.3). Annual soil CH_4 fluxes were positively correlated with C:N ratio and negatively correlated with Al saturation (Table 3.3). Considering only the area of mechanical disturbance (i.e., skidding trails, logging decks, and roads), soil CH_4 emissions were also positively correlated with SOC and ECEC ($r = 0.40$–0.52, $P < 0.01$–0.05, $n = 24$). Annual soil N_2O emissions did not correlate with any of the measured soil characteristics in Table S3.1.

Table 3.3

Spearman rank correlation coefficients between soil physical and biochemical characteristics in the top 10-cm depth and annual soil CO_2, N_2O and CH_4 fluxes across all strata in both logging systems in a Congo Basin rainforest of Cameroon.

Variables	Bulk density (g cm^{-3})	SOC[†] (kg m^{-2})	Total N (g m^{-2})	C:N ratio	Bray P[†] (g m^{-2})	ECEC[†] (mmol$_c$ kg^{-1})	Al saturation (%)
Soil CO_2 flux (Mg C ha^{-1} yr^{-1})	-0.82**	0.58**	0.69**	-0.07	0.74**	0.66**	0.01
Soil N_2O flux (kg N ha^{-1} yr^{-1})	0.01	0.13	0.06	0.23	-0.13	0.11	-0.16
Soil CH_4 flux (kg C ha^{-1} yr^{-1})	0.15	0.21	-0.01	0.51**	-0.13	-0.05	-0.32*

The correlation tests were conducted using the annual flux from each stratum of each replicate plot (2 logging systems × 4 replicate plots × 5 strata = $n = 40$).

[†] Soil organic C, Bray-extractable P, Effective cation exchange capacity.

** $P \leq 0.01$, * $P > 0.01 - \leq 0.05$.

3.5. Discussion

3.5.1. Soil CO_2 emissions

Soil CO_2 emissions from the undisturbed reference area (Table 3.1) were within the range reported for tropical rainforests on Ferralsol soils in Central and South America (92.7–228.3 mg C m^{-2} h^{-1}; Chambers et al., 2004; Davidson et al., 2004, 2000b; Keller et al., 2005; Matson et al., 2017; Schwendenmann et al., 2003; Sotta et al., 2006). The few studies in Africa conducted in forests and savannah with drier conditions (900–2050-mm annual rainfall) than our site have resulted lower soil CO_2 emissions (71.8–175.3 mg C m^{-2} h^{-1}; MacCarthy et al.,

2018; Wanyama et al., 2019; Werner et al., 2007) compared to our measurements (Table 3.1). The positive correlation between soil CO_2 emissions and WFPS across monthly measurements in the undisturbed reference area was similar to findings from earlier studies in (sub)tropical forests (e.g., Butterbach-Bahl et al., 2004; Liu et al., 2014; Matson et al., 2017; van Straaten et al., 2011). Under low WFPS, soil moisture can limit root and microbial activities, which is alleviated with increase in WFPS and thus increased soil respiration (e.g., Koehler et al., 2009a; Schwendenmann et al., 2003; van Straaten et al., 2011).

The decrease in soil CO_2 emissions from the disturbed area compared to undisturbed reference area (Table 3.1; Fig. 3.1a and b) corroborates previous studies in Malaysia (Mori et al., 2017; Yashiro et al., 2008) and France (Goutal et al., 2012). In contrast, Keller et al. (2005) found no alteration of soil CO_2 emissions following selective logging in Brazilian rainforests on clay Ferralsol and sandy loam Acrisol soils, which we attribute to moderate soil compaction at their skidding trails and logging decks. Moreover, Keller et al. (2005) did not include roads in their experimental design, which could underestimate the effect of selective logging, as we found highest changes in soil CO_2 emissions from roads. Increased WFPS in skidding trails, logging decks and roads (Fig. 3.2a and b; Table 3.2) can be explained by the increase in soil bulk density (Table S3.1), which had hampered water drainage following logging. The increase in soil temperature in logging decks and roads after logging is in agreement with previous studies (Yashiro et al., 2008; Zerva and Mencuccini, 2005) and can be related to the removal of the vegetation and litter layer that has exposed soils of these strata to direct sunlight. The removal of vegetation and litter layer could also explain the decrease in extractable mineral N in logging decks and roads, due to intensive decomposition of organic matter without fresh supply of litter to soils (Tchiofo Lontsi et al., 2019; Yashiro et al., 2008).

Similar to the undisturbed area, seasonal variation in soil CO_2 emissions across logging strata was driven by soil moisture content, as illustrated by the negative correlation with

WFPS during the wet season, when WFPS in the disturbed area ranged between 58 and 70% (Table 3.2). High WFPS might have restricted oxygen diffusion into the soil which, in turn, might have limited microbial activities; similarly, high WFPS could have hampered CO_2 transport within and from the soil surface (e.g., Davidson et al., 2000b; Koehler et al., 2009a; Sotta et al., 2004). The generally recognized parabolic relationship between soil CO_2 emissions and WFPS (Matson et al., 2017; Schwendenmann et al., 2003; Sotta et al., 2006; van Straaten et al., 2011) observed only when excluding logging decks and roads, suggests that soil compaction may have offset the seasonal regulation of soil CO_2 emissions by moisture content at logging decks and roads. The positive correlation found between soil CO_2 emissions and soil temperature in skidding trails, felling gaps and undisturbed area corroborates previous findings (e.g., Hassler et al., 2015; Liu et al., 2014; MacCarthy et al., 2018; Matson et al., 2017; Schwendenmann et al., 2003; Sheng et al., 2010; Sotta et al., 2006). Indeed, an increase in soil temperature under sufficient soil moisture can result in greater microbial activities leading to high CO_2 production in soils (Schlesinger and Bernhardt, 2013). However, in our study, range of soil temperature was narrow (23.9–27.0°C) and comparable between dry and wet seasons (Table 3.2), suggesting that the observed correlation is probably due to the difference in sunshine level and sampling time during the day, rather than a consistent seasonal trend in temperature. Extractable mineral N also influenced the temporal pattern of soil CO_2 emissions as shown by the positive correlation between them over the measurement period. Similar relationship has been reported from a tropical lowland forest in Panama and was related to the autocorrelation between soil moisture and extractable mineral N (Matson et al., 2017). At our study site however, there was no seasonal variation in extractable mineral N (Table 3.2) and its correlation with soil CO_2 emissions may reflect the effect of soil compaction in logging decks and roads, as discussed below, rather than a temporal regulation of extractable mineral N.

The large decreases in soil CO_2 emissions in roads, logging decks and skidding trails (Table 3.1) were, firstly, attributed to the decrease in soil organic matter in these highly disturbed strata, as indicated by the decreases in SOC and total N (Table S3.1) and exhibited in their positive correlations across all strata (Table 3.3). Previous studies in (sub)tropical ecosystems have reported a decrease in soil respiration following management practices that led to reduction in soil organic matter (e.g., Hassler et al., 2015; Liu et al., 2014; Sheng et al., 2010). Reduced amount of organic matter from the removal of the vegetation, surface litter and organic matter-rich topsoil in roads and logging decks might have decreased substrate for heterotrophic respiration, as heterotrophic respiration from litter can account 29–35% of soil respiration (Han et al., 2015; Liu et al., 2014; van Straaten et al., 2011). Also, removal of vegetation reduces input of organic material and eliminates root (autotrophic) respiration, which can account up to 50% of soil respiration (Mori et al., 2017; Schlesinger and Bernhardt, 2013; van Straaten et al., 2011). Secondly, the positive correlations of soil CO_2 emissions with Bray-extractable P and ECEC across strata suggest the regulation of nutrient availability, as P and base cations are commonly the limiting nutrient for decomposition activity in highly weathered Ferralsol soils (Kaspari et al., 2008). Extractable P and ECEC decrease with increasing degree of disturbance across strata (Table S3.1) due to decrease in organic matter (Tchiofo Lontsi et al., 2019), which also mirrored decrease in soil CO_2 emissions. Thirdly, the negative correlation found between soil CO_2 emissions and bulk density across strata (Table 3.3) was the result of soil compaction; soil bulk density increases in the skidding trails, logging decks and roads (Table S3.1; Tchiofo Lontsi et al., 2019). This increase in soil bulk density with increase degree of disturbance was the combined effect of heavy logging machinery and reduced organic matter (SOC and total N; Table S3.1; Tchiofo Lontsi et al., 2019), resulting in reduced soil CO_2 emissions in skidding trails, logging decks and roads compared to the undisturbed reference area (Table 3.1). Indeed, large soil bulk density (or reduced porosity)

could have restricted gas diffusion into and from the soil (Goutal et al., 2012; Yashiro et al., 2008), similarly to the effect of high WFPS discussed above.

In summary, the decrease in soil CO_2 emissions from the disturbed area compared to undisturbed area reflects the alteration of SOC and nutrient availability and the increase in WFPS following gradient of selective logging disturbance.

3.5.2. Soil N_2O emissions

Soil N_2O emissions from the undisturbed reference area (Table 3.1) were within the range of values reported for most tropical forests on Ferralsol soils across the humid tropics (3.4–29.7 µg N m^{-2} h^{-1}; Aini et al., 2015; Bauters et al., 2019; Castaldi et al., 2013; Matson et al., 2017; Verchot et al., 1999). In contrast, two other studies on Ferralsol soils in Central and South America reported soil N_2O emissions four times higher than our values, which may be attributed to the finer soil texture (i.e., clay-textured Ferralsol soils; Keller et al., 2005) compared to our sandy-loam soil texture or the sampling period (i.e., wet season only; Welch et al., 2019). Annual soil N_2O emissions from the undisturbed reference area (Table 3.1) were on the same order of magnitude with the average recently compiled for tropical forests on various soil types worldwide (2.0 kg N ha^{-1} yr^{-1}; van Lent et al., 2015), although the reported average mineral N content was seven times greater than extractable mineral N measured at our sites. Moreover, our values were within the range of soil N_2O emissions measured from some African rainforests (1.6 – 2.8 kg N ha^{-1} yr^{-1}; Bauters et al., 2019; Castaldi et al., 2013; Werner et al., 2007).

The positive correlation found between soil N_2O emissions and WFPS across monthly measurements in the undisturbed reference area is generally recognized in tropical forests (e.g., Corre et al., 2014; Davidson et al., 2004, 2000a; Gütlein et al., 2018; Keller et al., 2005;

Kiese et al., 2003; Koehler et al., 2009b; Matson et al., 2017; Raut et al., 2014; Werner et al., 2007). Indeed, high WFPS has created anaerobic conditions that were favorable to denitrification, the dominant process of N_2O production in tropical moist forest soils (e.g., Corre et al., 2014; Davidson et al., 2000a; Schlesinger and Bernhardt, 2013). We could not demonstrate a correlation between soil N_2O emissions from our undisturbed area and both NH_4^+ and NO_3^- contents that are also known to influence N_2O production in forest soils (e.g., Davidson et al., 2000a; Hassler et al., 2017; Matson et al., 2017; van Lent et al., 2015). Similar results were found in Australian tropical forests, wherein the authors suggested that the variability in N_2O emissions can be better explained by nitrification dynamics rather than actual soil mineral N content (Breuer et al., 2000; Kiese et al., 2003). The absence of correlation between extractable mineral N and soil N_2O emissions may be due to the low and narrow range of mineral N content at our undisturbed areas (3.0–5.7 mg N kg^{-1}; Table 3.2). Moreover, extractable mineral N did not change with season, which could explain comparable soil N_2O emissions from the undisturbed area in the dry and wet seasons.

The increase in soil N_2O emissions from the disturbed area following logging (83% relative to undisturbed reference area) corroborates previous studies in tropical forests (e.g., Keller et al., 2005; Yashiro et al., 2008). This change in soil N_2O emissions from the disturbed area was driven by high emissions from skidding trails (4.7 kg N ha^{-1} yr^{-1} on average in both logging systems). We expected similar changes in the other compacted areas (i.e., logging decks and roads) as result of low soil aeration from increased WFPS (Table 3.2; Fig. 3.2), but logging decks and roads had comparable emissions to the undisturbed reference area (Table 3.1). Soil aeration status appeared thus insufficient to explain the variability in soil N_2O emissions, as evident by the absence of correlation between soil bulk density and N_2O emissions across all strata (Table 3.3). It is possible that the removal of litter layer and topsoil from roads and logging decks has created substrate-limiting conditions for N_2O production,

which is evident by the lower mineral N content and total N in those strata compared to the reference (Tables 3.2 and S3.1). Leaf litter removal from the forest floor has also been shown to reduce soil N_2O emissions by approximately 42% relative to the control in a wet tropical forest of Costa Rica, as decomposition of the litter layer supplies substrate for nitrification (Wieder et al., 2011). The mineral N limitation to soil N_2O production in logging decks and roads is further illustrated by the positive correlation between soil NH_4^+ content and N_2O emissions, indicating that under very limited N conditions, a little increase in N availability could intensify N_2O production in soils. On the other hand, skidding trails had comparable N availability to the undisturbed reference area and felling gap (Tables 3.2 and S3.1), which might have resulted in comparable soil N_2O emissions. In contrast, soil N_2O emissions from the undisturbed reference area and felling gaps were lower than from skidding trails (Table 3.1). This suggests that lower WFPS in the undisturbed area and felling gaps may have favored the production of the more oxidized N gas (NO) rather than N_2O (Davidson et al., 2004, 2000a).

In summary, the combination of high WFPS (Table 3.2) and relatively high N availability (Tables 3.2 and S3.1) has favored larger N_2O emissions from skidding trails compared to the other strata. This is consistent with the hole-in-the-pipe model postulating that the first and second levels of control on soil N_2O fluxes are N availability and soil moisture content, respectively (Davidson et al., 2000a; Veldkamp et al., 2008).

3.5.3. Soil CH_4 fluxes

Undisturbed forest soils within our study area were sinks for atmospheric CH_4, and the uptake rate (Table 3.2) was within the range reported for African tropical forests on various soil types (-12.5 – -56.4 µg C m^{-2} h^{-1}; Gütlein et al., 2018; Kim et al., 2016; Wanyama et al.,

2019; Werner et al., 2007). In comparison with other studies from tropical lowland forests on Ferralsol soils, our values were higher than soil CH_4 uptake measured from Brazil and Panama ($-3.1 – -21.1$ µg C m^{-2} h^{-1}; Davidson et al., 2004; Keller et al., 2005; Matson et al., 2017). In contrast with the clay-texture of the above-mentioned Ferralsol soils, the coarse texture of our soils (sandy loam), has possibly enhanced the diffusion of atmospheric CH_4 into the soil, as well as oxygen that can be limiting for CH_4 oxidation (Veldkamp et al., 2013). The effect of soil aeration status on CH_4 fluxes was further illustrated in our study by the negative correlations between WFPS and soil CH_4 uptake, and between soil bulk density and soil CH_4 uptake (Section 3.4.1). Indeed, high WFPS or large soil bulk density may have affected gas diffusivity in the soil, thus reducing CH_4 availability for the activities of methane-oxidizing bacteria (methanotrophs) that are aerobic microorganisms (Raut et al., 2014; Schlesinger and Bernhardt, 2013). This control of soil moisture on soil CH_4 consumption conforms with previous studies conducted in tropical lowland forests (e.g., Davidson et al., 2004; Gütlein et al., 2018; Hassler et al., 2015; Keller et al., 2005; Matson et al., 2017; Veldkamp et al., 2013; Wanyama et al., 2019; Werner et al., 2007). The negative correlation between soil CH_4 fluxes and NO_3^- content in the undisturbed area (Section 3.4.1), indicates an increase in CH_4 uptake with increasing N availability. This is in agreement with previous studies that reported the limitation of CH_4 uptake by N availability (Hassler et al., 2015; Kiese et al., 2003; Matson et al., 2017; Veldkamp et al., 2013). Indeed, increasing N availability have either enhanced methanotrophic bacteria activities or increased their population leading to high CH_4 consumption in soils (Bodelier and Laanbroek, 2004; Veldkamp et al., 2013). Moreover, the availability of NO_3^- as electron acceptor could have reduced CH_4 production as end product of anaerobic decomposition of organic matter in soils, resulting in higher net CH_4 uptake (Topp and Pattey, 1997).

Soil in the disturbed strata changed from a sink to a net source of CH_4, except in felling gaps where CH_4 uptake was comparable to the undisturbed area (Table 3.1). Area-weighted average of CH_4 emissions from the disturbed strata (255.5 µg C m^{-2} h^{-1}) was very large and comparable in magnitude to the diffusive CH_4 flux measured at the surface of an Amazonian floodplain lake (257.4 µg C m^{-2} h^{-1}; Crill et al., 1988). Our results corroborate some previous studies that reported an increase in CH_4 production in forest soils following logging (Keller et al., 2005; Yashiro et al., 2008; Zerva and Mencuccini, 2005). On the other hand, no effect of logging on soil CH_4 fluxes was found in a Malaysian forest 14 years after logging (Mori et al., 2017). We speculate that either the time since logging may have allowed biogeochemical processes to recover, thus eliminating the effect of logging, or the experimental design (one-off sampling and no consideration of the spatial pattern of selective logging disturbances) may have failed to capture the spatial changes in soil CH_4 fluxes. By including the different disturbed strata in our design, we were able to observe various changes across a gradient of damage intensity from felling gaps (no change compared to the undisturbed reference area) to logging decks (high emissions) (Table 3.2).

We can attribute the net CH_4 emissions in the most disturbed strata (skidding trails, logging decks and roads) to the occurrence of anaerobic conditions due to soil compaction (high bulk density; Table S3.1), which directly affected WFPS (Fig. 3.2a and b; Table 3.2) and thus gas diffusion rate into the soil profile. Anoxic conditions in compacted soils have limited CH_4 consumption and enhanced CH_4 production by strictly anaerobic methanogenic archaea (Schlesinger and Bernhardt, 2013; Veldkamp et al., 2008; Verchot et al., 2000). The dominance of methanogenic over methanotrophic activities has resulted in net soil CH_4 emissions from the strata that experienced soil compaction. Furthermore, the net CH_4 uptake in dry season and net CH_4 emissions in wet season from skidding trails and roads suggest that CH_4 consumption dominated over production at lower WFPS despite soil compaction. This

highlights the dominance of WFPS as driver of CH_4 fluxes as previously reported (e.g., Matson et al., 2017; Veldkamp et al., 2013, 2008). There was however, no seasonal variation in CH_4 fluxes in logging decks, where ruts produced by heavy machinery were submerged during the wet season, which kept the soil moist even during the dry season. Therefore, due to anaerobic conditions in both dry and wet seasons, some areas of the logging deck became hot spots of CH_4 emissions as shown by the high variability among measurement chambers (large error bar; Fig. 3.1e and f).

Annual soil CH_4 fluxes across strata were controlled by SOC, ECEC, C:N ratio and Al saturation (Section 3.4.3; Table 3.3). The positive correlations of annual CH_4 fluxes with SOC and ECEC in the area of mechanical disturbance (skidding trails, logging decks and roads) depicted the control of soil fertility on CH_4 production or consumption as found in a previous study in tropical lowland forest in Panama (Matson et al., 2017). This indicates that under anaerobic conditions, SOC might be a limiting factor for CH_4 production in soil, as methanogenesis is a dominant pathway for the anaerobic decomposition of soil organic matter (Schlesinger and Bernhardt, 2013). Moreover, the positive correlation between soil CH_4 fluxes and C:N ratio (as indicator of N availability) across all strata (Table 3.3), may indicates the control of N availability over soil CH_4 uptake as explained above. The negative correlation between CH_4 fluxes and Al saturation across all strata may indicate an Al toxicity to soil methanogens, as also suggested for soil methanotrophs in previous studies (Hassler et al., 2015; Tamai et al., 2003).

3.6.   Conclusions

Soil CO_2, N_2O and CH_4 fluxes were all affected by selective logging. Ground disturbance associated with selective logging lowered annual soil CO_2 emissions by 28% and increased

soil N_2O emissions by 83%, when weighting the changes in the various disturbed strata by their respective areal coverage. A more significant change was observed for soil CH_4 as emissions from the disturbed area were more than seven times higher than CH_4 consumption in undisturbed area, thus comparable to CH_4 emissions from tropical wetlands. However, the disturbed strata represented only 4 to 5.2% of the logged area, which reduce considerably the changes in soil GHG fluxes when reported at the scale of the entire forest (i.e., a decrease of 1.3% and 36% in CO_2 emissions and CH_4 uptake respectively, and an increase of 3.3% in N_2O emissions). Meanwhile, increase in the area directly affected by selective logging may have a strong incidence on N_2O and CH_4 local budgets. The changes in GHG fluxes and soil factors (WFPS, soil temperature, extractable mineral N) were variable among the different disturbed strata. For example, soil GHG fluxes from the felling gaps did not differ from the undisturbed reference area, while N_2O emissions increased only in the skidding trails and CO_2 and CH_4 fluxes were strongly affected in skidding trails, logging decks, and roads. This supports our first hypothesis and highlights the importance of considering this spatial pattern of disturbance when assessing the impact of selective logging on forest soils. Changes in soil bulk density and WFPS that control gas diffusion into the soil, and in soil organic matter quantity and quality (e.g., SOC, total N, Bray-extractable P, ECEC, C:N, extractable mineral N) mainly explained the spatial and temporal variation in soil CO_2, N_2O and CH_4 fluxes, supporting our second hypothesis.

At our study site, CL and RIL used the same logging equipment and had similar effects on soil GHG fluxes. The difference between the two logging systems may therefore arise from the areal coverage of the disturbed strata, which can vary with the logging intensity and frequency. In our study site however, the extent of ground disturbance resulting from CL was atypically small, but frequent re-logging may not be avoided, as CL generally happened in unmanaged forests. However, because our study was done shortly after a single logging event,

it remains unclear how much cumulative area can be disturbed after consecutive loggings, and how long the logging-induced changes in soil GHG fluxes may last. Beyond these limitations, this study constitutes an interesting basis for further longitudinal research to improve knowledge on soil GHG flux changes following low-intensity selective logging in Congo Basin rainforests.

3.7.   Acknowledgments

3.8. References

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3.9. Appendix

Text S3.1. Soil sampling and analysis

Soil samples were taken from 12 randomly selected sampling points and mixed thoroughly to have one composite sample for each stratum (i.e., felling gap, skidding trail, logging deck, road, and undisturbed reference area) within each replicate plot (Fig. S3.1). Soil sampling was conducted between July and September 2016. Soil samples were taken at three depth intervals down to 50 cm, and we report in Table S1 the values for the top depth interval (0–10 cm). The soil samples were air-dried, sieved through 2-mm sieve, transported by air to Germany, and dried again at 40°C prior to analysis in the laboratory. Soil bulk density was determined using the core method (average of three measurements for each stratum in each replicate plot). Soil pH was measured from a 1:2.5 soil-to-distilled water ratio. Effective cation exchange capacity (ECEC) was determined by percolation with unbuffered 1 mol L^{-1} NH_4Cl and the exchanged cation concentrations (Mg, Ca, K, Na, Al, Fe, Mn) in percolates were measured using inductively coupled plasma – atomic emission spectrometer (ICP-AES; iCAP 6300 Duo VIEW ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). Base and Al saturations were calculated, respectively, as the percentage exchangeable bases (Mg, Ca, K and Na) and Al on ECEC. Subsamples of the composited soil samples were finely ground for measurement of total organic C, total N (using a CN analyser; Vario EL Cube, Elementar Analysis Systems GmbH, Hanau, Germany) and ^{15}N natural abundance signature (using isotope ratio mass spectrometry; Delta Plus, Finnigan MAT, Bremen, Germany). Phosphorus was extracted using the Bray 2 method and analyzed using the above-described ICP-AES.

Table S3.1

Soil physical and biochemical characteristics in the top 10-cm depth for the undisturbed reference area and disturbed strata within each logging system in a Congo Basin rainforest of Cameroon.

Logging system	Reference	Felling gap	Skidding trail	Logging deck	Road
Conventional					
Bulk density (g cm^{-3})	1.0 ± 0.0^b	1.0 ± 0.0^b	1.3 ± 0.0^{a,A}	1.4 ± 0.1^a	1.4 ± 0.1^a
pH (1:2.5 H$_2$O)	3.9 ± 0.1^{c,B}	3.9 ± 0.1^{c,B}	4.4 ± 0.1^b	4.7 ± 0.1^a	4.7 ± 0.1^{a,B}
ECEC (cmol$_c$ kg^{-1})	2.5 ± 0.1^{a,B}	2.3 ± 0.1^{ab}	1.7 ± 0.3^{abc}	1.5 ± 0.2^{bc}	1.2 ± 0.1^c
Base saturation (%)	15.2 ± 1.9^a	16.7 ± 4.5^a	26.2 ± 7.2^a	29.7 ± 6.8^a	38.9 ± 5.6^a
Aluminum saturation (%)	69.2 ± 2.1^a	64.2 ± 6.8^a	60.9 ± 6.4^a	61.0 ± 6.8^a	52.7 ± 4.9^a
^{15}N natural abundance (‰)	7.5 ± 0.3^{a,A}	6.8 ± 0.2^a	7.3 ± 0.4^a	8.0 ± 0.5^a	8.1 ± 0.2^a
C:N ratio	13.0 ± 0.4^a	13.3 ± 0.7^a	13.1 ± 0.9^a	13.7 ± 1.0^a	12.3 ± 0.7^a
Soil organic C (kg m^{-2})	1.7 ± 0.2^{a,B}	1.4 ± 0.1^{ab}	1.2 ± 0.2^{ab}	1.1 ± 0.2^{ab}	0.7 ± 0.0^b
Total N (g m^{-2})	129 ± 13^a	108 ± 13^{ab}	89 ± 14^{ab}	77 ± 17^{ab}	62 ± 3^b
Bray-extractable P (g m^{-2})	0.7 ± 0.1^a	0.7 ± 0.0^{ab}	0.4 ± 0.1^{bc}	0.3 ± 0.1^c	0.3 ± 0.1^c
Reduced-impact					
Bulk density (g cm^{-3})	1.0 ± 0.0^c	1.0 ± 0.1^c	1.2 ± 0.0^{b,B}	1.4 ± 0.1^a	1.4 ± 0.0^a
pH (1:2.5 H$_2$O)	4.2 ± 0.1^{b,A}	4.5 ± 0.1^{ab,A}	4.8 ± 0.2^a	4.9 ± 0.2^a	5.0 ± 0.0^{a,A}
ECEC (cmol$_c$ kg^{-1})	3.3 ± 0.2^{a,A}	2.6 ± 0.3^{ab}	2.8 ± 0.3^{ab}	1.9 ± 0.3^{bc}	1.4 ± 0.1^c
Base saturation (%)	20.1 ± 4.0^a	33.3 ± 7.2^a	44.5 ± 8.4^a	34.0 ± 8.5^a	30.3 ± 4.6^a
Aluminum saturation (%)	65.2 ± 5.4^a	54.8 ± 7.0^a	46.2 ± 6.6^a	57.9 ± 7.1^a	62.0 ± 4.5^a
^{15}N natural abundance (‰)	6.3 ± 0.2^{b,B}	7.1 ± 0.3^{ab}	6.5 ± 0.2^{ab}	7.2 ± 0.3^{ab}	7.6 ± 0.4^a
C:N ratio	14.1 ± 0.4^a	13.2 ± 0.4^a	15.4 ± 1.4^a	15.1 ± 1.8^a	13.7 ± 1.1^a
Soil organic C (kg m^{-2})	2.4 ± 0.2^{a,A}	1.9 ± 0.2^{ab}	2.1 ± 0.4^{ab}	1.4 ± 0.4^{ab}	0.9 ± 0.1^b
Total N (g m^{-2})	170 ± 17^a	146 ± 20^{ab}	134 ± 18^{abc}	84 ± 15^{bc}	67 ± 5^c
Bray-extractable P (g m^{-2})	0.6 ± 0.1^a	0.6 ± 0.1^a	0.4 ± 0.1^{ab}	0.2 ± 0.1^b	0.1 ± 0.0^b

Means (± SE, n = 4 plots) within a row followed by different lowercase letters indicate significant differences among strata within each logging system and uppercase letters indicate significant differences between logging systems within a stratum (ANOVA with Fisher's LSD test at $P \leq 0.05$).
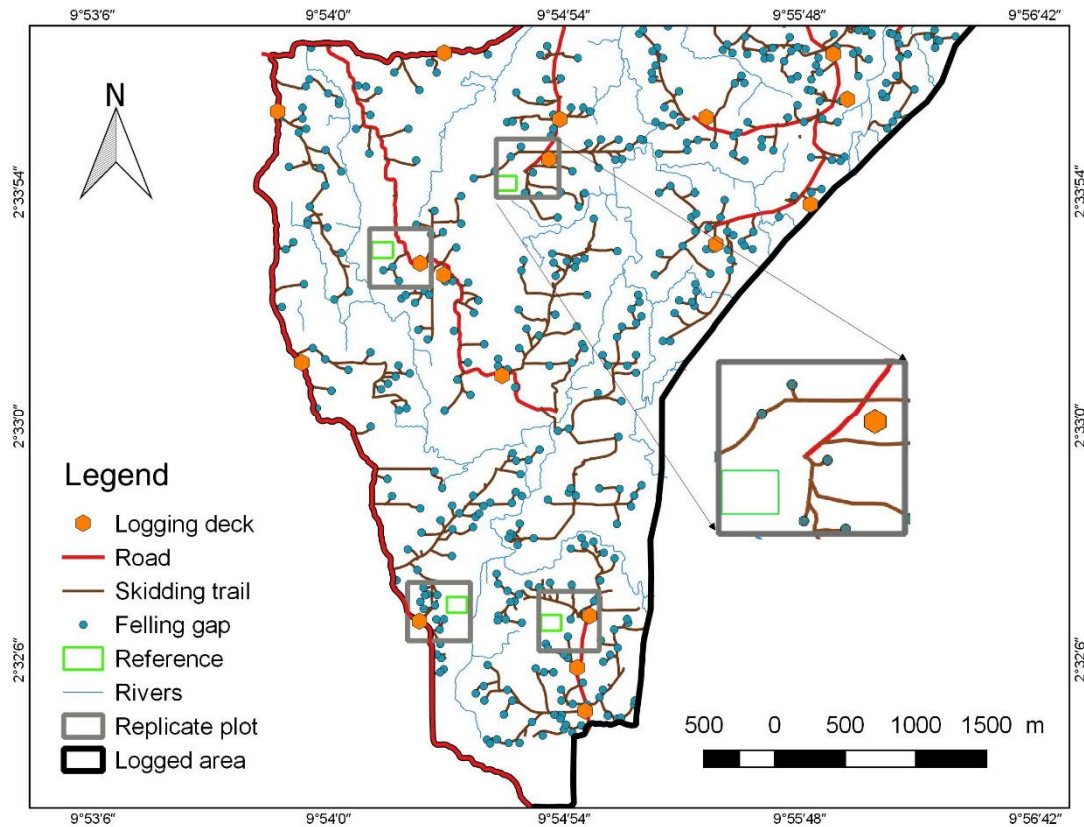
Fig. S3.1. Map illustrating the experimental design (e.g., reduced-impact logging site). Each of the four replicate plots had the four disturbed strata (road, logging deck, skidding trail and felling gap), and a corresponding undisturbed reference area, which was separated by at least 50 m distance from any disturbed stratum (Tchiofo Lontsi et al., 2019).

CHAPTER 4

SYNTHESIS

The overall goal of this study was to assess the impacts of selective logging on forest soils. We hypothesized that selective logging will affect soil characteristics and biochemical processes of GHG production/consumption, with higher impacts from conventional selective logging than reduced-impact logging. Although we could not find significant difference between the effects of both logging systems after a single logging event, we found significant change in major soil parameters, which followed the gradient of damage intensity within each logging system. In this chapter, I discuss the overall impact of selective logging at the scale of the entire forest area and simulate how an increase in logging intensity or frequency can affect ground damage and in turn logging-induced changes in soil characteristics and GHG fluxes.

4.1.   Ground damage from selective logging

The overall logging-induced changes in soil characteristics and GHG fluxes were related to the spatial extent of the disturbed strata. To estimate the disturbed area, we carried out a detailed ground mapping survey using GPS (Garmin), tape measure and GIS (see Chapter 2). All roads and logging decks were fully mapped, as they remained accessible for a long time after logging. We could verify our mapped skidding trails and felling gaps from the pre-planned logging map for RIL. Although the ground mapping used in the framework of our study was labor intensive and time consuming, it is the most affordable and recommended way to determine ground damage within a low disturbed forest (Ellis et al., 2016). For assessment of selective logging at larger scales (e.g., regional scale), remote sensing methods (e.g., visual identification of canopy gaps from low altitude aerial photographs or aerial LIDAR maps) may be more effective in quantifying the extent of logging disturbance, but are generally more costly. Moreover, remote sensing methods also require ground truth (i.e.,

similar to the method we used) for accuracy assessment, and aerial photographs or LIDAR maps should be taken shortly after the logging operations, as gap fraction decrease quickly with rapid colonization of disturbed area by early successional plant species (Asner et al., 2002; Ellis et al., 2016).

The generally recognized importance of reduced-impact logging (RIL) measures in minimizing the damage of selective logging to soils (Boltz et al., 2003; Holmes et al., 2002; Pereira Jr. et al., 2002) could not be demonstrated at our site. There are two possible reasons why the ground disturbance was comparable between CL and RIL. Firstly, the use of trained crew at the CL site resulted in low ground damage, even though there was no management plan. This suggests that crew training is a key component for reducing the environmental impacts of selective logging. Secondly, the logging intensity was comparable between the CL and RIL systems (0.2–0.3 tree ha^{-1}, equivalent to 2.75–2.78 m^3 ha^{-1}), which resulted in low ground disturbance in both systems. This indicates that in order to sufficiently assess the sustainability of a forest management system, critical attention should be paid to the logging intensity employed in each of the logging system, as the logging damage is strongly correlated to the number of harvested trees per unit of area (Durrieu de Madron et al., 2000; Pereira Jr. et al., 2002; Sist et al., 2003). Using the regression equation for mechanical disturbance (i.e., skidding trail, logging deck and road) (Fig. S2.1) and the felling gap size per harvested tree (Fig. 2.2), we can simulate the ground damage at various logging intensities. For example, an increase in logging intensity to one tree ha^{-1} (i.e., the generally reported maximum logging intensity in Cameroon; Cerutti et al., 2008) will result in a ground damage of 13.2 and 8.4% in CL and RIL respectively. Although these may represent overestimations as the mean felling gap size per harvested tree can decrease with increasing logging intensity due to possible overlapping gaps (Johns et al., 1996), they still illustrate that positive effects of RIL over CL practices are mostly realized at high logging intensities (Asner et al., 2009). Further research

effort to quantity selective-logging related damage is required to obtain a more reliable picture of ground area disturbed at various logging intensities in CL and RIL in this tropical ecosystem.

4.2.   Carbon losses and partial nutrient budget of selective logging

The extent of ground damage is insufficient in assessing the sustainability of a logging system, as the recovery of carbon to initial stocks is mainly driven by the level of damage to residual stand and soil fertility (Chazdon, 2003; Rutishauser et al., 2015). Indeed, recovery time can be used to assess whether the rotation period is long enough to prevent forest degradation. However our focus in this study was assessing logging damage to soils rather than to the residual stand. Nevertheless, we found that C exported in harvested timber (Table 2.3) represented only 0.8 and 0.6% of the pre-logging aboveground carbon stock in CL and RIL forests, respectively (Table 1.2). Estimations of biomass C emissions from selective logging (extracted timber + incidental damage to the residual stand + logging infrastructure) ranged from 0.99 to 2.65 Mg C m^{-3} of timber harvested in some Congo basin forests (Pearson et al., 2014; Umunay et al., 2019). Assuming the maximum reported emission factor of 2.65 Mg C m^{-3}, the low-intensity logging operations at our site will release approximately 7.29 and 7.37 Mg biomass C ha^{-1}, equivalent to 4.5 and 3.4% of the initial aboveground carbon stock in CL and RIL, respectively. If we assume a mean annual stand growth of 1.3 Mg C ha^{-1} following a logging damage to 15% of the residual stand (Djomo, 2010), the biomass C released by the logging operations can likely be rebuilt within six years. This suggests that low-intensity selective logging with sufficient rotation time can be sustainable in managing forest for aboveground C sequestration.

On average, 6.5, 0.2 and 1.4 kg ha^{-1} of N, P and K, respectively were exported in harvested timber (Table 2.3), which represented 0.1–1.3% of the nutrient stocks in the top 50-cm soil depth. Bauters et al. (2019) estimated N inputs of 18.2 kg N ha^{-1} yr^{-1} from bulk precipitation and N losses of 11.5 kg N ha^{-1} yr^{-1} from leaching in a Congo basin lowland rainforest on sandy loam Ferralsol soils. Using their estimates and our calculated gaseous N losses (soil N_2O emissions) of 1.6 to 2.1 kg N ha^{-1} yr^{-1} (Table 3.1), the partial N budget from our undisturbed forest area shows a net input of 4.6 kg N ha^{-1} yr^{-1}. We acknowledge, however, that this is a simplistic partial budget, because it does not include other N inputs such as N fixation and weathering, and outputs such as erosion and NO and N_2 emissions.

Combining the amount of N harvested with the changes in soil total N stocks in the top 50-cm depth (i.e., 59.3 kg N ha^{-1} on average for CL and RIL; Table 4.1), the forest will require at least 14 years to recover from the logging-induced N losses. As far as N balance in the system is concerned, this result suggests that low-intensity logging at our sites may be sustainable, as the amount of N lost from logging may be balanced by net inputs to plant-available pools over the rotation period of 30 years, especially in RIL. It is difficult to draw such conclusion for CL, as premature re-logging cannot be prevented.

4.3.   Logging-induced changes in soil biochemical characteristics

Our results showed that the highest changes in soil physical and biochemical characteristics happened in the logging decks and roads as a result of topsoil removal and subsoil compaction. Consequently, the impact of logging on soil characteristics can be substantial with an increase in the extent of these strata. We found overall reductions of 21–29% in the top 50-cm SOC, N and P stocks in the disturbed area relative to the undisturbed forest area (Fig. 2.3) within our logging sites. When taking the entire logged study site as a unit, a logging intensity of

≤ 0.3 tree ha^{-1} resulted in only 0.9–1.5% reductions in those element stocks compared to an unlogged forest (i.e., undisturbed reference area in this study). This implies that an increase in logging intensity to one tree ha^{-1}, with subsequent increase in disturbed area (see Section 4.1) will consequently intensify the reduction of SOC, N and P by a factor of 2 to 3 (i.e., a change of -1.8 to -3.8% relative to unlogged forest; Table 4.1).

Table 4.1

Relative change at the entire forest scale (mean ± SE; top 50-cm depth) in SOC, total N and Bray P with different logging intensities in conventional and reduced-impact logging.

Elements	Conventional logging		Reduced-impact logging	
	0.23 tree ha^{-1}	1 tree ha^{-1}	0.28 tree ha^{-1}	1 tree ha^{-1}
SOC (%)	-1.5 ± 0.4	-3.8 ± 1.1	-0.9 ± 0.3	-1.9 ± 0.7
Total N (%)	-1.3 ± 0.3	-3.3 ± 0.8	-0.8 ± 0.2	-1.8 ± 0.4
Bray P (%)	-1.2 ± 0.5	-3.2 ± 1.2	-1.2 ± 0.3	-2.4 ± 0.6

We used SOC as an example to simulate the effect of repeated logging within short period on soil biochemical characteristics. Assuming an annual recovery of 0.2% in SOC after selective logging (Chiti et al., 2016), logging-induced SOC losses may be recovered within seven to nine years (Fig. 4.1), if there is no additional logging in between. On the other hand, if there are repeated logging entries at 5-year intervals in the CL site with the same low logging intensity (i.e., resulting in the same extent of ground disturbance), SOC stock might never recover to the original level (Fig. 4.1). Our assumption of CL frequency is only one of many different scenarios, as logging may be less or more frequent, depending on timber market availability and governance issues. Similar changes may be observed for N and P, but data on their recovery rates were not available to run their scenarios.
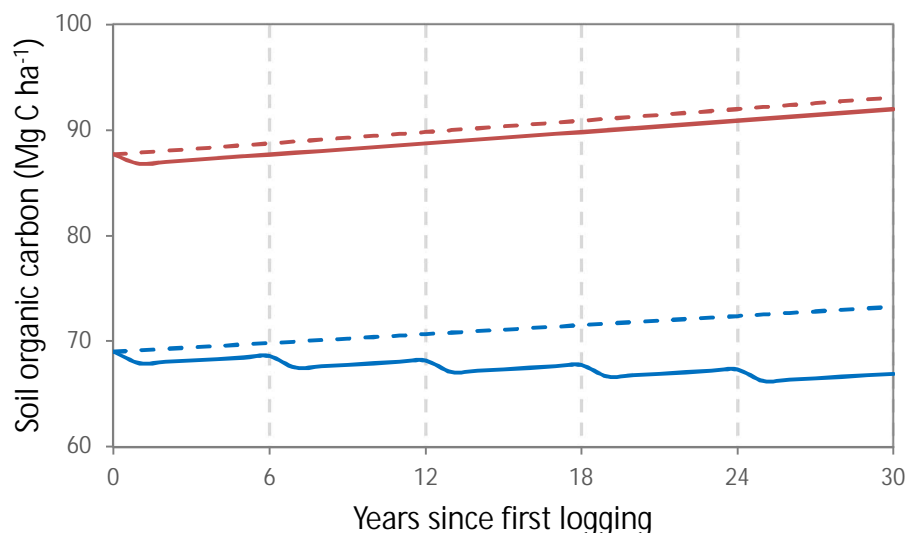
Fig. 4.1. Soil organic carbon recovery scenario in CL (blue) with repeated logging and in RIL (red) within a 30-year rotation time. Dash lines represent the SOC scenario without logging at CL and RIL sites and vertical dash lines correspond to consecutive loggings in CL.

4.4. Net soil global warming potential of selectively logged forests

The results presented in chapter 3 showed that soils in the disturbed areas had 28% less CO_2, 81% more N_2O and seven times larger CH_4 emissions than the undisturbed areas, which acted as CH_4 sinks. At the scale of the entire logged forest, logging induced a decrease of 1.3 and 36% in soil CO_2 emissions and CH_4 uptake, while it increased soil N_2O emissions by 3.3% relative to undisturbed forest across CL and RIL (Table 4.2). Assuming a logging intensity of one tree ha^{-1}, we estimate that logging may reduce soil CO_2 emissions by 3%, while reducing the capacity of the forest to remove CH_4 from the atmosphere by ~87%, as well as increase soil N_2O emissions by about 8% (Table 4.2). This implies that a subsequent increase in logging intensity may change the soils of these logged forests from a net sink to a net CH_4 source. It is worth stressing that these changes are sensitive to the area of ground disturbance with increasing logging intensities. It highlights once more the need for further research to properly

estimate the change in ground disturbance with increasing logging intensity in these highly diverse forests.

We used the concept of global warming potential (GWP) to assess the cumulative impact of selective logging on the three soil GHG fluxes measured in this study. We used the conversion factors of 1 for CO_2, 265 for N_2O and 28 for CH_4 (IPCC, 2014) to estimate the net GWP of our forest soils in CO_2 equivalent (CO_2 eq.) over a 100-year time scale. Annual soil CO_2 equivalent emission from the undisturbed area (Table 4.2) was within the range reported for natural ecosystems in Africa (11.7–121.3 Mg CO_2 eq. ha^{-1} yr^{-1}; Kim et al., 2016). Annual net soil GWP was lower in logged forest compared to unlogged forest (Table 4.2), due to decreased soil CO_2 emissions from areas of soil compaction. However, the reduced soil CO_2 emissions from the disturbed area should not be interpreted as reduced net ecosystem CO_2 emissions, as vegetation that uptake CO_2 has been removed from the disturbed area. Assuming a net primary productivity of 17.6 Mg C ha^{-1} yr^{-1} reported for a west African forest (Morel et al., 2019), there will be a net ecosystem uptake of 4.8–5.7 Mg CO_2-C ha^{-1} yr^{-1} in the undisturbed forest area, while the net ecosystem emission in the disturbed area will remain 12.9–12.5 Mg CO_2-C ha^{-1} yr^{-1} in RIL and CL. Consequently, the net ecosystem uptake will decrease to 4.0–4.7 Mg CO_2-C ha^{-1} yr^{-1} in the logged forest under actual logging intensities of 0.2–0.3 tree ha^{-1}. Albeit stem CH_4 emissions in the undisturbed area (i.e., 0.5 µg C m^{-2} stem $hour^{-1}$; Iddris et al, unpublished data), net CH_4 uptake remains lower in logged forest (1.0–3.1 kg CH_4-C ha^{-1} yr^{-1} in RIL and CL) compared to unlogged forest (2.4–4.2 kg CH_4-C ha^{-1} yr^{-1} in RIL and CL). Similarly, assuming stem N_2O emissions of 1.5 µg N m^{-2} stem $hour^{-1}$ in the undisturbed forest (Iddris et al, unpublished data), total N_2O emissions will remain slightly higher in RIL logged forest (2.1 kg N_2O-N ha^{-1} yr^{-1}) than unlogged forest (2.0 kg N_2O-N ha^{-1} yr^{-1}). These results illustrate that selective logging may increase GHG fluxes.

Table 4.2

Annual soil GHG fluxes (mean ± SE) and equivalent global warming potential from undisturbed forest and forest logged at different intensities.

Logging intensity (tree ha^{-1})	Soil CO$_2$ (Mg C ha^{-1} yr^{-1})	Soil N$_2$O (kg N ha^{-1} yr^{-1})	Soil CH$_4$ (kg C ha^{-1} yr^{-1})	Soil GWP (Mg CO$_2$eq. ha^{-1} yr^{-1})
Conventional logging				
Undisturbed	17.0 ± 1.0	2.1 ± 0.4	-4.3 ± 0.8	63.05
0.2 (observed)	16.7 ± 1.0	2.1 ± 0.4	-3.3 ± 1.2	61.98
1.0 (simulated)	16.4 ± 1.0	2.1 ± 0.3	-1.6 ± 1.9	60.95
Reduced-impact logging				
Undisturbed	18.3 ± 1.1	1.6 ± 0.3	-2.5 ± 0.8	67.67
0.3 (observed)	18.1 ± 1.0	1.7 ± 0.3	-1.1 ± 0.4	67.03
1.0 (simulated)	17.8 ± 1.0	1.9 ± 0.3	0.4 ± 1.5	66.07

4.5. Implication for forest management

Our study showed that selective logging influences soil nutrient levels and soil GHG fluxes, though the magnitude of changes depends on the extend of disturbed area. Concessionaires can reduce avoidable logging damage through proper planning of logging infrastructures. They can focus for example on reducing the number and area of logging decks that generate greatest changes in soil characteristics (e.g., reduction in soil fertility; Table 2.2) and GHG fluxes (especially CH$_4$ emissions; Table 3.1). Creating less logging decks may increase the area of skidding trail, thus increasing N$_2$O emissions from the disturbed area, as these emissions were highest from the skidding trail (Table 3.1). To avoid the impacts of ground skidding such as soil compaction from using bulldozers, we suggest the use of long-line cable

winching, instead of creating skidding trails up to the felled-tree stumps. Winching technology has been shown to significantly reduce skidding trail density and consequently skidding impacts on forest soils (Griscom et al., 2019).

According to the forest regulations of Cameroon, concessionaires are required to close the logging roads and reforest the disturbed strata. Our results showed that regeneration is likely to be constrained by reduced soil fertility in logging decks and roads (Chazdon, 2003). Restoration of soil fertility may therefore be a prerequisite for forest recovery in those strata, and may require human assistance. Alternatively, regeneration efforts can be focused in skidding trails and felling gaps where nutrient availability coupled with light intensity (i.e., in felling gap) can promote seedling growth. Concessionaires can therefore replace pioneer species (e.g. *Musanga cecropioides*) that generally colonize felling gaps by commercial tree species. For future logging events, concessionaires should avoid opening new roads and logging decks and prioritize the use of existing ones from previous logging. Unfortunately, this is unlikely to happen in CL since there is no logging map and logging is not always done by the same company.

4.6.   Outlook

Our study provides data on the spatial variability of short-term effect of low-intensity selective logging on soil characteristics and GHG fluxes and on nutrient export with harvested timber. It can be used to estimate the response of forests on similar soil conditions to selective logging operations, provided the areal coverages of all disturbed strata are known. These data can serve as baseline information for longitudinal studies to assess long-term effects of selective logging on forest soils. To improve our understanding of the topics discussed in this dissertation, further research is needed to study among others:

- the effect of increased logging intensity on ground-disturbance;

- the cumulative effect of consecutive logging on soil characteristics and corresponding GHG fluxes;

- post-logging recovery rates of soil element stocks;

- the duration of logging perturbations to soil GHG fluxes.

4.7.   References

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To God be the glory!

THESIS DECLARATION

I, Rodine Tchiofo Lontsi, hereby declare that I have composed the present thesis independently using no other sources and resources than those stated. I have accepted the assistance of third parties only in a scope that is scientifically justifiable and compliant with the legal statutes of the examinations. In particular, I have completed all parts of the dissertation myself; I have neither, nor will I, accept unauthorised outside assistance either free of charge or subject to a fee.

Furthermore, I have not applied for an equivalent doctoral examination elsewhere and submitted the present thesis as a whole or in parts at another university.

Göttingen, December 2019   _____

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