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# Management intensity controls soil $N_2O$ fluxes in an Afromontane ecosystem

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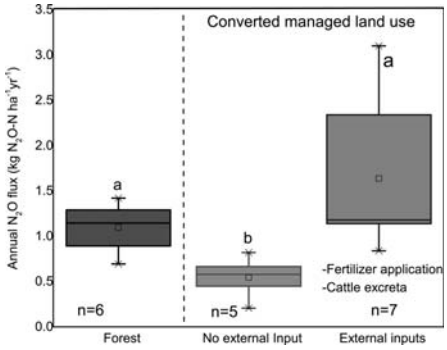
## HIGHLIGHTS

- Unique monitoring study of nitrous oxide fluxes from a tropical montane forest and adjacent land uses in Kenya.
- Annual nitrous oxide fluxes from forest soils are lower than those from lowland tropical forests in Africa.
- Land management intensity influences soil nitrous oxide fluxes more than land use alone because of high variability.
- Total inorganic N intensity explains variations in annual soil $N_2O$ fluxes.
- Management masks effect of soil properties on soil $N_2O$ fluxes.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Studies that quantify nitrous oxide ($N_2O$) fluxes from African tropical forests and adjacent managed land uses are scarce. The expansion of smallholder agriculture and commercial agriculture into the Mau forest, the largest montane forest in Kenya, has caused large-scale land use change over the last decades. We measured annual soil $N_2O$ fluxes between August 2015 and July 2016 from natural forests and compared them to the $N_2O$ fluxes from land either managed by smallholder farmers for grazing and tea production, or commercial tea and eucalyptus plantations ($n = 18$). Air samples from 5 pooled static chambers were collected between 8:00 am and 11:30 am and used within each plot to calculate the gas flux rates. Annual soil $N_2O$ fluxes ranged between 0.2 and 2.9 kg N $ha^{-1}$ $yr^{-1}$ at smallholder sites and 0.6–1.7 kg N $ha^{-1}$ $yr^{-1}$ at the commercial agriculture sites, with no difference between land uses ($p = 0.98$ and $p = 0.18$, respectively). There was marked variation within land uses and, in particular, within those managed by smallholder farmers where management was also highly variable. Plots receiving fertilizer applications and those with high densities of livestock showed the highest $N_2O$ fluxes ($1.6 \pm 0.3$ kg $N_2O$-N $ha^{-1}$ $yr^{-1}$, $n = 7$) followed by natural forests ($1.1 \pm 0.1$ kg $N_2O$-N $ha^{-1}$ $yr^{-1}$, $n = 6$); although these were not significantly different ($p = 0.19$). Significantly lower fluxes ($0.5 \pm 0.1$ kg N $ha^{-1}$ $yr^{-1}$, $p < 0.01$, $n = 5$) were found on plots that received little or no inputs. Daily soil $N_2O$ flux rates were not correlated with concurrent measurements of water filled pore space (WFPS), soil temperature or inorganic nitrogen (IN) concentrations.

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However, IN intensity, a measure of exposure of soil microbes (in both time and magnitude) to IN concentrations was strongly correlated with annual soil $N_2O$ fluxes.



## 1. Introduction

Nitrous oxide ($N_2O$) is a potent greenhouse gas (GHG), estimated to contribute about 6% to anthropogenic climate forcing (Blanco et al., 2014). The atmospheric $N_2O$ concentration has increased from 270 ppbv during the pre-industrial era to approximately 320 ppbv, mainly due to stimulated soil $N_2O$ emissions following the use of increasing amounts of reactive N synthesized via the Haber-Bosch process for crop production (Parkin et al., 2012). While agricultural soils are considered major $N_2O$ sources primarily due to fertilizer application, tropical forest soils are also a major natural $N_2O$ source because of often high soil N availability and environmentally favorable conditions for $N_2O$ production (Fowler et al., 2009; Werner et al., 2007a).

In soils, $N_2O$ is mainly produced through two microbial, enzyme-mediated processes: nitrification (autotrophic and heterotrophic) and denitrification (Butterbach-Bahl et al., 2013; Davidson et al., 2000), although other production pathways such as nitrifier-denifrication (Kool et al., 2010) and dissimilatory nitrate reduction to ammonia (Silver et al., 2001) have also been reported. Autotrophic nitrification is enhanced by oxygen availability, moderate water content (approximately 60% water filled pore space WFPS), ammonium ($NH_4^+$-N) availability, temperature >5 °C and soil pH >5. Heterotrophic nitrification requires organic carbon (C), $NH_4^+$-N supply and occurs in acidic soils (Wood, 1990; Zaman et al., 2012). Denitrification, an anaerobic microbial process where nitrogen oxides are used as alternative terminal electron acceptors instead of $O_2$, is driven by high soil water content (above 60% WFPS) as this hampers $O_2$ diffusion and results in creation of soil anaerobiosis. Besides the availability of nitrate ($NO_3^-$) and nitrite ($NO_2^-$), denitrification also requires the availability of easily degradable C substrates. Several studies have observed a linear relationship between $NO_3^-$-N pools and soil $N_2O$ fluxes (Groffman et al., 2000; Schelde et al., 2012). However, at higher levels of $NO_3^-$-N (>0.4 μg $NO_3^-$-N $g^{-1}$) the $N_2O$ flux yield by denitrification often decreases (Gelfand et al., 2016; Schelde et al., 2012) as C substrate availability might become the rate limiting factor. Both nitrification and denitrification therefore, are influenced by the size of inorganic-N pools in the soil, and these pools depend on N turnover through mineralization and soil amendments such as fertilizers and livestock excreta.

Nitrification and denitrification have been linked to $N_2O$ fluxes through a conceptual "hole in the pipe" model (Davidson et al., 2000) that links fluxes to the "size of the pipe" (i.e. the amount of N that is nitrified and denitrified), and the "size of the holes" (i.e. the $N_2O$ losses from each process). Typically, this model relates the hole-size to soil water content, which controls the anaerobic status of the soil through its effect on gas diffusion. However, prediction of $N_2O$ fluxes based on simultaneously observed environmental factors and substrate concentrations ($NH_4^+$-N and $NO_3^-$-N) shows very weak to no correlations in most studies (Gelfand et al., 2016; Maharjan and Venterea, 2013; Veldkamp et al., 2008; Wolf et al., 2011), partly because of complex interactions between drivers and temporal variation in soil moisture. Mixed evidence has been reported with strong correlations between cumulative $N_2O$ and cumulative $NO_3^-$, referred to as nitrate intensity (Burton et al., 2008), however another study found no relationship between either nitrate or ammonium intensity and annual $N_2O$ flux but did find a strong correlation with nitrite intensity (Maharjan and Venterea, 2013).

Measurements of GHG fluxes from agricultural and natural ecosystems in Africa are limited (Kim et al., 2016; van Lent et al., 2015). Recently, some studies have measured soil $N_2O$ emissions from African tropical forests covering lowland (Castaldi et al., 2013; Gharahi Ghehi et al., 2013; Werner et al., 2007b), and montane (Gütlein et al., 2017) forests. However, these studies cover mostly a few weeks and thus do not capture seasonal variability in fluxes (Werner et al., 2007b). Also, the focus of these studies has been on natural forests and not necessarily on the succeeding land uses. Only a few studies, (e.g. Gütlein et al., 2017; Arias-Navarro et al., 2017b) have attempted to fill this data gap and have studied GHG fluxes from tropical montane forests and compared those to agricultural land uses. However, the latter study is an incubation study with intact soil cores and applied regression analysis using observed changes in soil moisture to calculate annual fluxes.

In the tropics, primarily in the Brazilian Amazon and Sumatra, conversion of natural forest to agricultural land use has been shown to elevate soil $N_2O$ emission for a short period after which emissions become lower or equal to the original forest (Melillo et al., 2001; van Lent et al., 2015; Verchot et al., 2006). In land uses where inorganic fertilizers and organic/manure inputs were used, soil $N_2O$ emissions were often greater than those from the original forest soils (Katayanagi et al., 2008; Lin et al., 2012; Veldkamp et al., 2008).

Land use change involves changes in vegetation type and management practices that may cause changes in soil organic stocks and their quality (Metcalfe et al., 2011), soil microbial communities and microclimate modification (i.e. soil temperature and water content), all of which could influence GHG fluxes (Gates, 2012). The Mau forest is the largest contiguous montane forest in Kenya (Wass, 1995). Land use change in this forest has occurred rapidly since the 1960s driven by the expansion of smallholder agriculture and by commercial agriculture. While tea plantations replaced forests >50 years ago, smallholder agriculture, primarily for grazing or for small-scale tea plantations, continues to drive forest loss. Within large tea estates, the main land uses are either tea or eucalyptus and cypress plantations, with the wood used as fuel for the boilers to run the tea processing plants. On both the small and large-scale farms, tea fields are typically fertilized with NPK (26% N, 5% $P_2O_5$ and 5% $K_2O$) compound fertilizer once or twice a year suggesting that emissions from these fields could be higher than emissions from the natural forests.

The aim of this study was to quantify annual soil $N_2O$ emissions from a tropical montane forest and compare these to the annual soil $N_2O$ emissions from converted land uses: grazing land, tea in smallholder agriculture, tea in commercial plantations and eucalyptus plantations. We also examined mineral nitrogen availability, soil pH, soil temperature and soil water content to explain spatial changes in soil $N_2O$ fluxes. We hypothesized that tea fields and grazing lands would have higher soil $N_2O$ fluxes compared to natural forest and eucalyptus plantations due to fertilizer application and animal excreta deposition. In addition, we hypothesized that natural forests would have greater soil $N_2O$ emissions than the eucalyptus plantations.

## 2. Experimental methods and design

### 2.1. Study sites

This study was carried out in the South West (SW) Mau forest of Kenya in East Africa. The Mau forest is a tropical montane forest, with high rates of deforestation (Baldyga et al., 2008). Overall, forest cover was reduced from 520,000 ha to 340,000 ha between 1986 and 2009 (Hesslerova and Pokorny, 2010), while between the 1990s and early 2000s the forest area of the SW Mau decreased from 84,000 to 60,000 ha (Kinyanjui, 2009). The vegetation in the SW Mau is classified as afro-montane mixed forest with broad-leafed species such as *Polyscias fulva* (Hiern.Harms), *Prunus Africana* (Hook. f Kalkman), *Macaranga*