

## Elevated methane concentrations in trees of an upland forest

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Received 15 May 2012; revised 1 July 2012; accepted 3 July 2012; published 9 August 2012.

[1] There is intense debate about whether terrestrial vegetation contributes substantially to global methane emissions. Although trees may act as a conduit for methane release from soils to atmosphere, the debate centers on whether vegetation directly produces methane by an uncharacterized, abiotic mechanism. A second mechanism of direct methane production in plants occurs when methanogens – microorganisms in the domain Archaea – colonize the wood of living trees. In the debate this biotic mechanism has largely been ignored, yet conditions that promote anaerobic activity in living wood, and hence potentially methane production, are prevalent across forests. We find average, growing season, trunk-gas methane concentrations  $>15,000 \mu\text{L}\cdot\text{L}^{-1}$  in common, temperate-forest species. In upland habitat (where soils are not a significant methane source), concentrations are 2.3-times greater than in lowland areas, and wood cores produce methane in anaerobic, lab-assays. Emission rate estimates from our upland site are  $52 \pm 9.5 \text{ ng CH}_4 \text{ m}^{-2} \text{ s}^{-1}$ ; rates that are of a similar magnitude to the soil methane sink in temperate forest, and equivalent in global warming potential to  $\sim 18\%$  of the carbon likely sequestered by this forest. Microbial infection of one of the largest, biogenic sinks for carbon dioxide, living trees, might result in substantial, biogenic production of methane. **Citation:** Covey, K. R., S. A. Wood, R. J. Warren II, X. Lee, and M. A. Bradford (2012), Elevated methane concentrations in trees of an upland forest, *Geophys. Res. Lett.*, 39, L15705, doi:10.1029/2012GL052361.

### 1. Introduction

[2] Containing more than 75% of terrestrial carbon, forests are globally-important sinks and stores for carbon [Houghton, 2007]. Because in upland soils the activities of methane-consuming bacteria (i.e. methanotrophs) generally dominate those of methane-producing archaea (methanogens), forests are also considered sinks for atmospheric methane ( $\text{CH}_4$ ) [Conrad, 2009]. However, recent work indicates that forests may be producing and emitting huge quantities of  $\text{CH}_4$ . Using remotely-sensed data Frankenberg *et al.* [2005] showed unexpectedly high concentrations of

$\text{CH}_4$  over the tropics. On-the-ground measurements of  $\text{CH}_4$  flux suggested the emissions might come from a novel, aerobic mechanism through which live vegetation and litter in forests act as a methane source [Keppler *et al.*, 2006]. More than 30 studies have attempted to explain, measure, and verify observations of  $\text{CH}_4$  production via this pathway and two recent reviews conclude that the phenomenon does occur [Bruhn *et al.*, 2012; Keppler *et al.*, 2009]. Yet there is still no definitive confirmation or rejection that forests on well-drained soils are a significant  $\text{CH}_4$  source [Anderson, 2010].

[3] If vegetation does produce  $\text{CH}_4$ , the magnitude of emissions may contribute significantly to global fluxes. At the low end, the U.S. Environmental Protection Agency estimates emissions from vegetation of  $20 \text{ Tg}\cdot\text{yr}^{-1}$ ; roughly equivalent to the global warming potential (GWP) of  $\text{CO}_2$  released through U.S. residential use of fossil fuels. At the high end, emissions from vegetation may be  $60 \text{ Tg}\cdot\text{yr}^{-1}$ ; approximately equal to the GWP of fossil-fuel  $\text{CO}_2$  emissions from combined U.S. industrial, residential and commercial sources [Anderson *et al.*, 2010; Environmental Protection Agency, 2011]. To quantify  $\text{CH}_4$  emissions from vegetation, two general methods have been used: bottom-up approaches where flux measurements for individual plants or field plots are extrapolated to regional or global scales [Keppler *et al.*, 2006; Kirschbaum *et al.*, 2006; Parsons *et al.*, 2006]; and top-down approaches that identify “missing”  $\text{CH}_4$  from global models and reconcile it with proposed emissions from land [Aydin *et al.*, 2011; Bousquet *et al.*, 2006; Frankenberg *et al.*, 2005; Houweling, 1999; Kai *et al.*, 2011]. The validity of global  $\text{CH}_4$  emission estimates from both approaches hinges on whether vegetation directly produces  $\text{CH}_4$  [Nisbet *et al.*, 2009]. Certainly, in water-inundated soils, trees can act as a conduit for  $\text{CH}_4$  release from soils to the atmosphere [Rice *et al.*, 2010; Rusch and Rennenberg, 1998; Terazawa *et al.*, 2007]. And although UV light does seem responsible for direct  $\text{CH}_4$  production from vegetation, this aerobic mechanism remains uncharacterized [Bruhn *et al.*, 2012; Keppler *et al.*, 2009]. This has led to omission of  $\text{CH}_4$  production from living plants as a source in recent, global  $\text{CH}_4$  budgets [Dlugokencky *et al.*, 2011]. However, a second mechanism of direct  $\text{CH}_4$  production from living plants exists: the archaeal methanogens colonizing the wood of trees [Schink and Ward, 1984; Van Der Kamp *et al.*, 1979; Xu and Leininger, 2001; Zeikus and Ward, 1974; Zeikus and Henning, 1975].

[4] In the natural world, disease and decay commonly occur together. Decay of organic matter produces gases such as  $\text{CO}_2$  and  $\text{CH}_4$ , and hence is a fundamental determinant of global biogeochemical cycling rates and atmospheric chemistry. Concentrations of  $\text{CH}_4$  as high as 60% have been found in tree boles [Bushong, 1907]. At least one

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**Table 1.** Species-Level Differences in CH<sub>4</sub> Production, Timber Volume and Susceptibility to Fungal Decay

Species	Mean Production Potential ( $\mu\text{g CH}_4 \text{ m}^{-3} \text{ s}^{-1}$ )	Percent of Cores Showing CH <sub>4</sub> Production	Range of Production <sup>a</sup> ( $\mu\text{g CH}_4 \text{ m}^{-3} \text{ s}^{-1}$ )	Standing Bole Volume ( $\text{m}^3 \text{ ha}^{-1}$ )	Percentage of Volume Lost to Heart Rot <sup>b</sup>
<i>Pinus strobus</i> L. (eastern white pine)	0.067	40	.161–.175	8.6	5%
<i>Tsuga canadensis</i> L. (eastern hemlock)	0.115	40	.190–.383	11.5	12%
<i>Quercus rubra</i> L. (red oak)	0.107	60	.165–.192	28.2	19%
<i>Betula lenta</i> L. (black birch)	1.555	80	.191–6.884	8.6	21%
<i>Acer rubrum</i> L. (red maple)	1.239	60	.236–3.522	8.6	21%

<sup>a</sup>Range of production includes only those samples demonstrating measureable production of CH<sub>4</sub>. Reported means are for all samples.

<sup>b</sup>Wagener and Davidson [1954].

source of high trunk-gas CH<sub>4</sub> concentrations has been known for more than forty years: bacterial infection of heartwood (i.e. non-living tissue that primarily accumulates in trunks as trees age). This infection promotes wetwood and, with it, production of CH<sub>4</sub> through classical methanogenesis [Zeikus and Ward, 1974]. Wetwood CH<sub>4</sub> production has not been quantified for its contribution to global CH<sub>4</sub> emissions [Conrad, 2009]. Similarly, neither have contributions from heart rot – the decay of heartwood instead caused by fungal infection [Boyce, 1961] – which also promotes anaerobic decay and colonization by methanogens, but is much more prevalent in living trees. Indeed, in temperate forests ~20% of the commercial timber harvest, for genera such as oaks and maples, is lost to fungal decay [Wagener and Davidson, 1954]. Notably, symptoms of heart rot are often not outwardly visible for standing trees [Zillgitt and Gevorkiantz, 1948] and anaerobes can be active before decay is measurable [Shortle et al., 1978; Wilcox, 1970].

[5] In low O<sub>2</sub> and high CO<sub>2</sub> environments, such as those in tree trunks [Teskey et al., 2008], aerobic heart-rot fungi are incapable of completing their metabolic processes [Jensen, 1967; Schmidt, 2006]. This incomplete fermentation provides substrates suitable for use by bacteria and archaea. In turn, these bacteria and archaea accelerate fungal growth by removing the waste products of fungal metabolism and by enriching the wood substrate through N-fixation [Beckmann et al., 2011]. These syntrophic (i.e. “feeding together”) consortia are capable of breaking down complex biopolymers that individual organisms cannot digest [Bryant et al., 1967]. Such consortia are known to degrade wood and produce CH<sub>4</sub> in ruminant animals [Bauchop, 1981; Joblin and Naylor, 1989], in digesters [Zinder, 1993] and in timbers stored under conditions similar to those found inside living trees [Beckmann et al., 2011; Krüger et al., 2008]. Even in predominately aerobic environments, fungal metabolism can lead to anaerobic microsites and the formation of large quantities of CH<sub>4</sub> by archaea [Reith et al., 2002].

[6] Given the expectation of widespread and abundant fungal infection of living wood, we selected individuals in lowland and upland habitat of six tree species that vary in their vulnerability to heart rot [Scheffer, 1966] and that commonly occur in temperate forest. We reasoned that if trees primarily serve as conduits of CH<sub>4</sub> release from soils to the atmosphere, then we should only observe elevated CH<sub>4</sub> concentrations in tree trunks where the soil might be a significant CH<sub>4</sub> source (i.e. lowland habitat). A second source in these lowland habitats would be wetwood, which would also yield elevated CH<sub>4</sub> concentrations in trees. In contrast, in upland habitat the soil is a CH<sub>4</sub> sink and wetwood is rare. We therefore reasoned that elevated CH<sub>4</sub> concentrations in tree

trunks of species known to be susceptible to fungal-mediated heart rot would suggest an abundant and widespread CH<sub>4</sub> production source in living trees. Using trunk-gas CH<sub>4</sub> concentrations, and lab-based CH<sub>4</sub> production potentials from wood samples, we provide estimates of emissions from living trees of CH<sub>4</sub> produced by the microbial consortia that occur with heart rot.

## 2. Methods

### 2.1. Field and Laboratory Measurements

[7] Because decay is more likely to be found in larger and older trees [Berry and Beaton, 1972; Browne, 1956; Zillgitt and Gevorkiantz, 1948], we selected 58 trees with diameters at breast height (dbh; 1.3 m) >25 cm (indicative in our region of mature, canopy trees in middle-aged stands). Trees were selected by order-of-encounter, stratified across six species (Table 1), in lowland and well-drained upland habitat at Yale-Myers Forest, Connecticut, USA (Lat. 41°56'15" Long. –72°10'45"). Stands were of similar age-class (~80–100 years), and representative of the oak-dominated hardwood forest type common to the eastern U.S. [Meyer and Plusnin, 1945].

[8] To determine *in situ* trunk-gas CH<sub>4</sub> concentrations, prior to (April) and post (July) leaf-out in 2011, trees were drilled horizontally at breast height to center with a 5/16" drill bit (Speedbor, Irwin, Huntersville, NC, USA) and immediately plugged with an 8-mm stopper (SubaSeal, Sigma-Aldrich, St. Louis, MO, USA). A 50-mL gas-syringe (SGE, Ringwood, AU) was inserted through the SubaSeal and into the cavity to remove 50 mL of trunk-gas from each tree, 15 mL of which was injected into a vacuum-sealed 12-mL pre-evacuated sample vial (Exetainer, Labco, High Wycombe, UK) and 0.2 mL analyzed by gas chromatography on an FID Gas Chromatograph (310C, SRI, Menlo Park, CA, USA) equipped with a 1-m silica-gel column, with helium as a carrier gas and an oven temperature of 40°C.

[9] We evaluated trunk-gas CH<sub>4</sub> concentrations using General Linear Mixed Models to assess effect of habitat and time of year, with tree species as a random factor, therefore accounting for spatial and temporal associations in our sampling design. To discern species-level differences, we used ANOVA to assess time-of-year by species effects. All models were run in the statistical freeware R [R Development Core Team, 2010].

[10] To confirm that trunk-wood had the potential to produce CH<sub>4</sub>, in October 2011 we removed bark-to-pith increment cores from the same trees, sectioned them to fit in 37-mL anaerobic bottles, flushed them with 50 mL of 100%-N<sub>2</sub> and returned them to the laboratory within 12 h of collection. Headspace were flushed again with N<sub>2</sub> for 3 min

at 1 L·min<sup>-1</sup> and incubated for 12 h at 20°C, after which a 15 mL sample was withdrawn and measured in the same manner as the trunk-gas samples. These lab assays only provide evidence for CH<sub>4</sub> production from trunk wood and rate comparisons are probably not reliable. For example, CH<sub>4</sub> production can drop rapidly following disturbance of methanogenic communities and similar assays with wetwood-infected materials show that N<sub>2</sub> assays underestimate production potentials by a factor of ~3 [Mukhin and Voronin, 2011; Zeikus and Ward, 1974].

## 2.2. Scaling to Field Rates for Upland Forest

[11] Past work investigating tree-mediated CH<sub>4</sub> transport from anoxic soils has demonstrated that CH<sub>4</sub> diffuses through bark [Gauci et al., 2010; Pulliam, 1992; Rusch and Rennenberg, 1998; Terazawa et al., 2007], and studies of other trunk gases have shown bark flux rates are positively and linearly related to trunk gas concentrations [Steppe et al., 2007]. Given these relationships, we estimated *in situ* bark effluxes for our upland site from trunk-gas CH<sub>4</sub> concentrations and radial diffusivity in wood. Ignoring longitudinal diffusion, we obtain the diffusion equation in the cylinder coordinate as:

$$F = -f\rho\rho_a D \frac{r_1}{r_2} \left( \frac{\partial\omega}{\partial r} \right)_{r_2}$$

where  $F$  is the radial diffusion flux,  $f$  is a radial diffusivity scale factor (= 0.017) [Zohoun et al., 2003] similar to the tortuosity factor used to describe gas diffusion in soils,  $\rho$  is air filled-porosity estimated to be 0.07 according to the water content reported for wet wood [Nord-Larsen et al., 2011],  $\rho_a$  is air density,  $D$  is CH<sub>4</sub> diffusivity in ambient air (= 0.21 cm<sup>2</sup> s<sup>-1</sup>, [Massman, 1998],  $\omega$  is CH<sub>4</sub> molar mixing ratio, and  $r_1$  and  $r_2$  are the radius of the heartwood and tree trunk, respectively. We used the mean tree radius ( $r_2 = 23.5$  cm) and assumed the central half of this radius was heartwood ( $r_1 = 11.7$  cm), giving 11.7 cm of trunk wood for CH<sub>4</sub> diffusion. The mixing ratio gradient at  $r_1$  was estimated from the observed difference in the mixing ratio between the trunk air and ambient air. The flux computed from the above equation has the dimensions of  $\mu\text{g CH}_4 \text{ m}^{-2} \text{ s}^{-1}$  (unit surface area of the tree trunk) and was converted to  $\mu\text{g CH}_4 \text{ m}^{-3} \text{ s}^{-1}$  (unit wood volume) for the purpose of upscaling. Mean calculated flux rates were scaled to per hectare field rates using standing live bole volume (Table 1) estimated from region and species-specific volume equations [Meyer and Kienholz, 1944]. Volumes estimates were based on randomized variable-radius plot sampling ( $n = 5$  plots) in our upland site using a factor 10 basal area prism (Cruise Master, Forestry Suppliers, Jackson, MS, USA). Emissions from those species not sampled (11% of total volume), were scaled using the mean concentration of all sampled species.

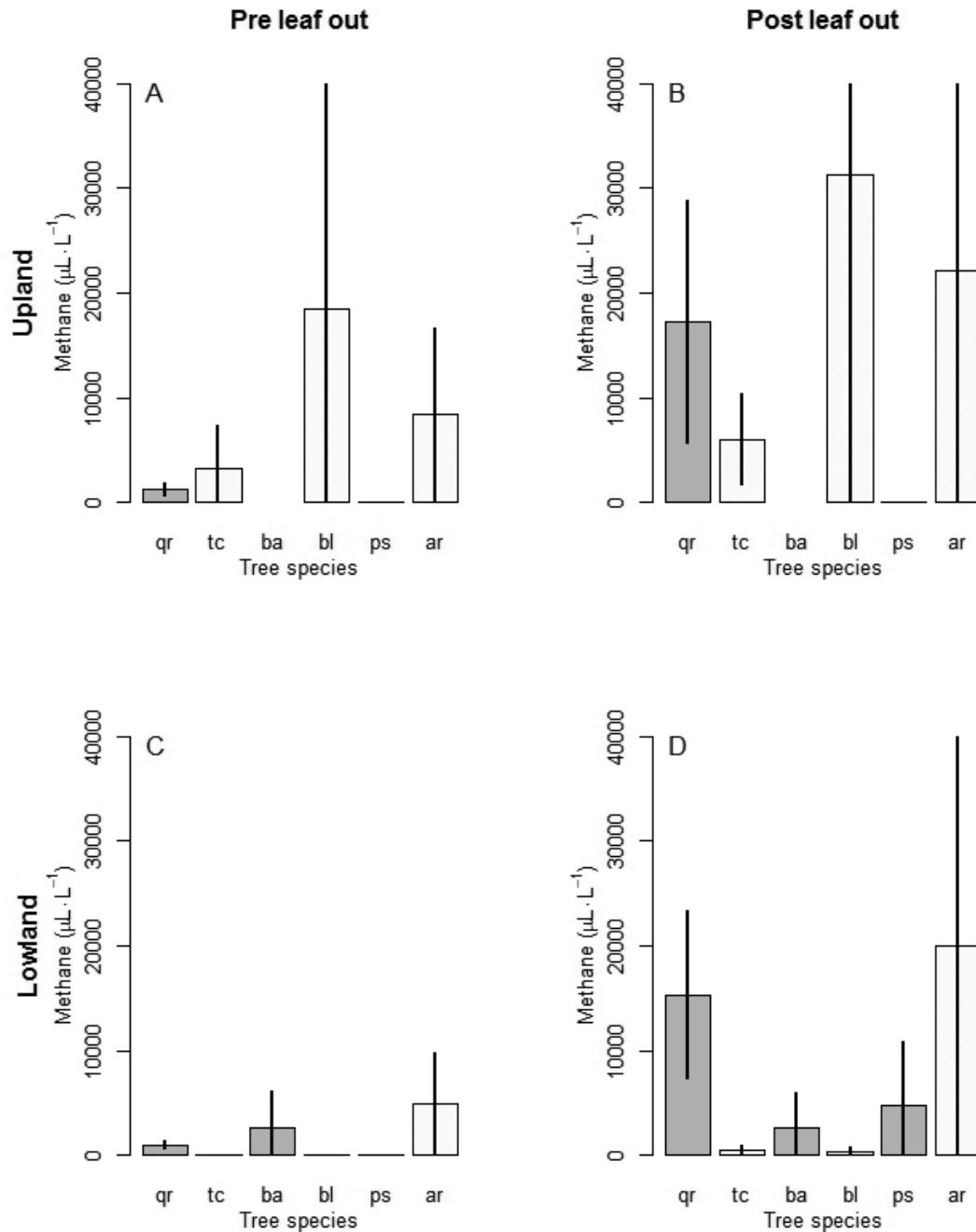
## 3. Results and Discussion

[12] Our data suggest that trees – by supporting environments suitable for classical methanogenesis – could make upland forests significant contributors to global CH<sub>4</sub> emissions. At our upland site, trunk-gas CH<sub>4</sub> concentrations were as high as ~80,000 times atmospheric, and mean growing-season concentrations in the three species (red maple, red oak, black birch; Table 1) known to be most susceptible to

heart rot were above 15,000  $\mu\text{L L}^{-1}$  (Figure 1), leading to species-level differences in CH<sub>4</sub> trunk-gas concentrations ( $P < 0.01$ ). A similar trend across species was present in lab assays with black birch and red maple producing more consistently and at higher rates than the two conifer species studied (eastern white pine, eastern hemlock). These assays confirm the production of significant quantities of methane in outwardly healthy tree wood under anaerobic headspace. The large variation present may be due to high spatial variability of production within individuals or could stem from the sensitivity of microbial communities to disturbance. We assessed increment cores and found visible decay was not correlated with trunk-gas CH<sub>4</sub> concentrations and/or production potentials. This lack of correlation is not surprising, and we would not expect a direct correlation with rot for several reasons. For example, trees may respond to injury and microbial infection by generating anoxia in trunk wood, favoring methanogens but generating little visible decay [Shortle, 1979]. It is at these decay frontiers where microsites favoring methanogenesis are most likely to occur and advanced decay would likely reduce or shut-down methanogenesis because it increases permeability facilitating O<sub>2</sub> diffusion [Schwarze, 2007; Sorz and Hietz, 2006]. Notably, in the upland site, we had a red maple individual in which the center was hollow and trunk CH<sub>4</sub> concentrations approximated those in ambient air.

[13] Trunk CH<sub>4</sub> concentrations were 2.3-times greater ( $P = 0.06$ ) in trees of upland habitat, where heart rot is expected to be more prevalent than at lowland sites [Basham, 1973]. More importantly, upland soils typically consume rather than produce CH<sub>4</sub> [Bradford et al., 2001], suggesting that the bulk of the trunk CH<sub>4</sub> was produced internally and did not accumulate via soil-tree diffusion pathways [Rice et al., 2010]. Further support for this interpretation was provided when we returned to the upland site in February 2012 and, in 10 red oak individuals, found CH<sub>4</sub> concentrations lower at the trunk base (5 cm above the soil) than at 1.3 m (i.e. dbh; mean difference = 9,551  $\mu\text{L L}^{-1}$ ,  $P = 0.08$ ). This pattern is opposite to that observed when trees are functioning as conduits for release of CH<sub>4</sub> produced in soils [Rusch and Rennenberg, 1998]. Lastly, relative CH<sub>4</sub> accumulation in individual trees appeared consistent across seasons, being temporally correlated ( $\log_{10}$  [pre-leaf out CH<sub>4</sub>] = 0.7556 \*  $\log_{10}$  [post-leaf out CH<sub>4</sub>] + 0.8574,  $r^2 = 0.43$ ,  $P < 0.05$ ) and on average 3.1-times greater ( $P = 0.083$ ) in summer than spring, following the expected temperature sensitivity of methanogenesis [Conrad, 2009].

[14] Mean *in situ* diffusion fluxes across all species, estimated from the trunk-gas CH<sub>4</sub> concentrations and lateral gas diffusivity in wood, were  $7.1 \pm 1.3 \mu\text{g CH}_4 \text{ m}^{-3} \text{ s}^{-1}$  (mean  $\pm$  SE, per unit wood volume). Scaling field-diffusion flux estimates to local field rates, emission rates are  $52 \pm 9.5 \text{ ng CH}_4 \text{ m}^{-2} \text{ s}^{-1}$ , respectively. These emissions have a GWP equal to ~18% of the carbon these stands likely sequester per annum [Law et al., 2002], and are of a similar magnitude to annual mean CH<sub>4</sub> consumption rates by bacteria (i.e. methanotrophs) in temperate forest upland soils [Bradford et al., 2001]. The resulting net fluxes are therefore below the minimum detection limit for eddy covariance [Kroon et al., 2010], providing a parsimonious explanation as to why such measurement approaches have not previously identified this potential source.



**Figure 1.** Trunk-gas methane concentrations in *Quercus rubra* (qr: red oak), *Tsuga canadensis* (tc: eastern hemlock), *Betula alleghaniensis* (ba: yellow birch), *Betula lenta* (bl: black birch), *Pinus strobus* (ps: eastern white pine) and *Acer rubrum* (ar: red maple), in lowland and well-drained upland habitat. Ambient air concentrations at 1.3-m height were consistently below  $2 \mu\text{L L}^{-1}$ . Values are means  $\pm$  95% CIs;  $n = 5$  in upland and 5 or 6 in lowland.

[15] The production of  $\text{CH}_4$  by the heart rot pathway questions whether upland forests can be considered a net sink for atmospheric  $\text{CH}_4$  (through soil consumption). To answer this question, further work is required to refine our  $\text{CH}_4$  emission estimates and determine their contribution to the global  $\text{CH}_4$  budget. Specifically, our work was conducted in intermediate-aged stands in temperate woodland on  $\sim 60$  trees, and applying these rates to large areas would

assume similarity across disparate forest types. Instead, susceptibility to fungal decay varies by species, site, age-class, past management regimes, and between and within individuals [Wagener and Davidson, 1954]. For example, within individuals, heart rot often starts at the base or “butt” of the tree [Krause and Gagnon, 2005; Wagener and Davidson, 1954] but we observed higher  $\text{CH}_4$  concentrations at 1.3 m height, as opposed to 5 cm above the soil. This

is probably because trunk-gas O<sub>2</sub> concentrations are highest at the base and lowest mid-way up the stem [Eklund, 2000], potentially decoupling extent of heart rot from CH<sub>4</sub> production rates. At the stand level, tree age and successional status will likely impact CH<sub>4</sub> emissions. For example, age is closely related to the likelihood of heart rot, and older stands generally have higher standing-wood volumes [Hennon, 1995], meaning older stands will presumably have higher CH<sub>4</sub> emissions. Although stand-level emissions may increase with age (as long as trunk decay remains enclosed within the tree), heart rot is also known to affect younger trees, particularly when subjected to managements such as coppice. Lastly, positive relationships between temperature, moisture and decay rates could result in a latitudinal gradient in forest CH<sub>4</sub> production, with tropical biomes – where heart rot can cause as much as 30% loss in merchantable timber volume [Grogan and Schulze, 2008] – having greater emissions than temperate and boreal forests. Such a latitudinal pattern would be consistent with observed atmospheric CH<sub>4</sub> concentrations, which are highest above the moist tropics [Frankenberg et al., 2005].

[16] Our data, uncertainties in global CH<sub>4</sub>-emission sources [Heimann, 2011], the ubiquity of heart rot [Wagener and Davidson, 1954], and the fact CH<sub>4</sub> production from heart-wood occurs through a known, biological mechanism [Beckmann et al., 2011; Zeikus and Ward, 1974], makes plausible globally-significant production of CH<sub>4</sub> from living trees via the heart rot pathway. To gain precise global-scale estimates of CH<sub>4</sub> production by living trees through this pathway will require on-the-ground assessments of individual trees across all major forest types, managements and age classes. Until such work is conducted, uncertainties in the size of CH<sub>4</sub> emission sources, and in explanations of temporal and spatial dynamics in global, atmospheric CH<sub>4</sub> concentrations, are unlikely to be reduced.

#### 4. Conclusion

[17] The potential for disease to regulate biogeochemical cycling is recognized [Hudson, 2006], but disease of one of the largest, biogenic sinks for carbon – the wood of living trees – has received little to no consideration in how it might affect atmospheric chemistry and associated climate change. Our data reveal trunk-gas CH<sub>4</sub> concentrations many times atmospheric on both lowland and upland sites. The highest concentrations were found for the upland site, and in species known to be susceptible to heart rot, suggesting this disease as the pathway of CH<sub>4</sub> production. The common infection of trees by heart rot fungus, and associated bacteria and archaea, has long been a concern of commercial forestry. These findings suggest decay in living trees is also important to biogeochemists and atmospheric scientists seeking to understand the role of forests in the global CH<sub>4</sub> budget.

[18] **Acknowledgments.** We thank Mikailah McKee, Matthew Fried, Erin Raboin, Jonathan Loevner, Jonathan Sullivan, and Victoria Lockhart for field assistance, the Director and Managers at the Yale Myers Forest for logistical support, Helmut Ernstberger for technical assistance, and Chad Oliver and the other members of the Greeley LMS Lab for comments on the project. Josh Schimel provided critical feedback on an earlier version of the manuscript. Funding was from the Yale Institute for Biospheric Studies and School of Forestry and Environmental Studies.

[19] The Editor thanks Timo Vesala and an anonymous reviewer for their assistance in evaluating this paper.

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