

**C and GHG balances
in an age-sequence
of temperate pine
plantations**

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Carbon and greenhouse gas balances in an age-sequence of temperate pine plantations

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1 Introduction

The global temperature increase over the past century has been attributed to increasing concentrations of atmospheric greenhouse gases (GHGs) such as carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O), among others, coupled with human activities (e.g. fossil fuel burning, deforestation) and natural processes in terrestrial ecosystems (Houghton et al., 1998; IPCC, 2007, 2013; Schimel, 1995; Schulze et al., 2009). Much effort has since been put into mitigating the atmospheric GHG concentrations through sequestration and/or their emission reduction strategies through land use management activities (Montzka et al., 2011; Nabuurs et al., 2007; Smith et al., 2008).

Among land use types, forest ecosystems have the strongest carbon (C) sink potential with estimated stocks of 100 to 160 Gt C in biomass and soil in temperate regions alone (Dixon et al., 1994; Goodale et al., 2002; Pan et al., 2011; Reich, 2011). Moreover, much of the current CO₂ sink strength of the Northern Hemisphere has been attributed to re-growth and afforestation of former agricultural land in the US and Canada (Pan et al., 2011; Reich, 2011). Therefore, further afforestation of former low productive agricultural lands has been proposed to help offset the increasing concentration of atmospheric CO₂ (Bárcena et al., 2014; Brown, 2002; Goodale et al., 2002; Nabuurs et al., 2007; Niu and Duiker, 2006).

Afforestation of former agricultural lands and associated changes in land management practices, however, does not only result in accumulation of C into biomass but may also alter soil and micro-climatic conditions. Tree root development, litter-fall, canopy shading and cessation of N-fertilizer application, for instance, may trigger changes in physical, biogeochemical and hydrological properties of the soil which may affect the net exchange of CH₄ and N₂O (Ball et al., 2007; Christiansen and Gundersen, 2011; Christiansen et al., 2012; Gundersen et al., 2012; Peichl et al., 2010b; Priemé et al., 1997; Smith et al., 2003; Ullah and Moore, 2011), as well as the cycling of dissolved organic carbon (DOC) (Camino-Serrano et al., 2014; Gielen et al., 2011; Peichl

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et al., 2007; Rosenqvist et al., 2010). Such alterations might modify the net ecosystem production (NEP) and the GHG balance (Luyssaert et al., 2010, 2012; Schulze et al., 2009, 2010). Moreover, since CH₄ and N₂O have a 25 and 298 times greater global warming potential (GWP) relative to CO₂ on molar basis over 100 years (IPCC, 2007), afforestation effects on the exchange of these GHGs might be even more pronounced with respect to the net radiative forcing (Inatomi et al., 2010; Montzka et al., 2011; Schulze et al., 2009).

The impacts of afforestation on ecosystem C pools and GHG fluxes may also change with stand age over time, including short-term (< 10 years; Bjarnadottir et al., 2009; Don et al., 2009; Zona et al., 2013), intermediate (~ 10 to 50 years; Ball et al., 2007; Christiansen and Gundersen, 2011) and long-term (> 50 years; Coursolle et al., 2012; Hiltbrunner et al., 2012; Priemé et al., 1997) effects. In addition, site quality (which includes all environmental factors influencing tree growth and thus biomass production and decomposition) might exert a strong control on the forest CO₂ exchange (Fernández-Martínez et al., 2014; McLeod and Running, 1988; Peichl et al., 2010a; Vanninen et al., 1996). Changes in the magnitude of the CO₂ exchange subsequently has implications for the relative contribution of the individual C and GHG fluxes to the total C and GHG budgets following afforestation. Thus, knowledge on the changes over the entire life span of the afforested stand and on the initial site conditions and quality are required to understand the overall implications of such land use practices for the C and GHG balances at ecosystem and regional scales.

To date, few attempts have been made to estimate the full forest C and GHG balance by including measurements of all relevant fluxes of CO₂, CH₄, N₂O, and DOC. Ball et al. (2007) estimated that the combined contribution of CH₄ and N₂O to the total GHG balance was 6–7% in maturing (20–30 years old) coniferous forests. In a young poplar plantation, the contribution of CH₄ and N₂O was reported to account for 50% of the GHG balance (Zona et al., 2013). Using a process-based model, Inatomi et al. (2010) estimated the GHG balance of a 50 year-old temperate broadleaved forest at 10 t CO₂ ha⁻¹ yr⁻¹ by including all three GHGs (CO₂, CH₄, N₂O), with a small (< 2%)

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contribution from CH₄ and N₂O fluxes. On a larger regional scale, Tian et al. (2010) suggested in their model study that emissions of CH₄ and N₂O from terrestrial ecosystems may offset about two thirds of the land CO₂ sink over the North American continent. However, DOC fluxes were not included in these studies. Gielen et al. (2011) reported that the lateral export of DOC accounted for 11 % of the NEP in forests. Thus, these previous studies indicate substantial contributions from non-CO₂ carbon and GHG fluxes and the need for including these exchanges when estimating and/or modeling full forest C and GHG budgets.

In this study, we estimated the C and GHG balances of four afforested pine stands ranging from 7 to 70 years of age. The main objectives were (i) to determine the effects of stand age and site quality on the forest C and GHG balances and (ii) to investigate the contribution of non-CO₂ fluxes (i.e., CH₄, N₂O and DOC) to the forest C and GHG balances.

2 Material and methods

2.1 Study sites

This study was conducted in four eastern white pine (*Pinus strobus* L.) forests, which are part of the Turkey Point Flux Station located on the north shore of Lake Erie in Southern Ontario, Canada (42°71' N, 80°36' W). The region has a temperate climate with a 30-year mean annual temperature of 8.0 °C and an annual precipitation of 1036 mm (Environment Canada Norms from 1981–2010 at Delhi, Ontario). The two older forests were planted in 1939 (TP39; 70 years-old in 2009) and 1974 (TP74; 35 years-old in 2009) on cleared oak-savannah land, while the two younger stands were established on former agricultural lands in 1989 (TP89; 20 years-old in 2009) and in 2002 (TP02; 7 years-old in 2009). Although all four stands grow on generally similar sandy and dry soils, TP89 and TP02 are characterized by greater site quality compared to the two older sites (Table 1). The higher site quality at the two younger stands might

result from post-agricultural remnants of nutrients (e.g. elevated soil nitrogen and pH, see Table 1). In addition, high site productivity at the 20 year-old site, TP89, is further due to a shallow ground water table that allows trees to have continuous access to water (Peichl et al., 2010a). Each site has a tower instrumented for eddy covariance flux and meteorological measurements. A suite of standard instruments for soil temperature and soil moisture measurements at several depths are also installed at each site. More details of site-specific instrumentation stand and soil characteristics are given in previous studies (Arain and Restrepo-Coupe, 2005; Khomik et al., 2010; McLaren et al., 2008; Peichl and Arain, 2006; Peichl et al., 2010a).

2.2 Biometric estimates of C pools and fluxes

Above- and belowground C pools in forest biomass and soil were determined in permanent sample plots established at each site in 2004 following the Canadian National Forest Inventory Protocol (NFI, 2003) as described in detail in Peichl and Arain (2006) and Peichl et al. (2010c). Using data from these NFI inventory plots, biomass and detritus pools of canopy and understorey trees, ground vegetation, and forest floor (LFH-layer) were determined each year in autumn. Soil C and fine root biomass pools were determined from soil coring in 2004. Litter-fall was collected seasonally to bi-annually using litter traps. Forest floor litter decomposition rates were determined over four years (from autumn 2004 to autumn 2008) at the three older sites. Litter bags with 1 mm mesh size were filled with 10 g of air-dried pine needles and 20 bags were retrieved annually to determine the mass loss.

Net primary production (NPP) was determined from the annual changes in living biomass (ΔB_L) and detritus (ΔB_D) pools based on annual inventory and litter-fall data combined with site-specific allometric biomass equations (Peichl and Arain, 2007) (Eq. 1). Soil respiration (R_S) was measured from 2004–2008 using a LiCor LI-6400 portable chamber system (Khomik et al., 2006, 2009). Soil heterotrophic respiration (RH_S) was computed using a site-specific soil respiration model derived from trenched plot respiration data and further partitioned into heterotrophic respiration from min-

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eral soil (RH_M) and litter layer (RH_{LFH}) (Eq. 2) (Peichl et al., 2010b, c). Heterotrophic respiration from aboveground woody debris (RH_{WD}) was derived from woody debris decomposition rates (Black et al., 2007; Law et al., 2001) and woody debris pools and added to RH_S to obtain total heterotrophic respiration (RH) (Eq. 3). Autotrophic root respiration (RA_R) was determined from the difference between R_S and RH_S (Eq. 4), while autotrophic aboveground canopy respiration (RA_C) was derived by subtracting RA_R from RA (Eq. 5), with RA being the difference between biometric estimates of NPP and GPP (Peichl et al., 2010c). The biometric estimate of RE was calculated as the sum of the individual respiration terms (Eq. 6).

$$NPP = \Delta B_L + \Delta B_D \quad (1)$$

$$RH_S = RH_M + RH_{LFH} \quad (2)$$

$$RH = RH_S + RH_{WD} \quad (3)$$

$$RA_R = R_S - RH_S \quad (4)$$

$$RA_C = RA - RA_R \quad (5)$$

$$RE = RH_S + RH_{WD} + RA_R + RA_C \quad (6)$$

The biometric estimates of individual component fluxes were used to demonstrate their within-ecosystem partitioning, whereas total RE and GPP were further combined with EC-derived estimates on the stand level as described further below (Sect. 2.4).

2.3 DOC export

The DOC export as the difference between input via precipitation and soil leaching below 1 m depth was estimated for the snow free periods (April to December) of 2004 and 2005. Precipitation was collected in plastic buckets equipped with a funnel on top of meteorological towers at bi-weekly to monthly intervals and analyzed for its DOC concentration using a Shimadzu 5050 Analyzer. Soil water samples from 1 m depth were taken to estimate the null-point DOC concentration (DOC_{np}) as indicator of the

mean RE and GPP fluxes therefore do not exactly match the sum of their individual biometric component flux estimates.

2.4.2 CH₄ and N₂O fluxes

Fluxes of CH₄ and N₂O were determined at monthly intervals at all four sites from April to December of 2006 and 2007 using the static chambers as described by Peichl et al. (2010b). Briefly, air samples (20 mL) were withdrawn using syringes at 0, 30, 60, and 90 min through a plastic tube inserted into evacuated glass vials (13 mL) equipped with a gray butyl septum in the field and analyzed within one week for CH₄ concentrations using a Shimadzu Mini Gas Chromatograph (GC) (Shimadzu Scientific Instruments, Columbia, ML, USA) equipped with a methanizer and a Flame Ionization Detector (FID) detector and for N₂O concentrations using a Shimadzu 14-A GC (Shimadzu Scientific Instruments, Columbia, ML, USA) equipped with an electron capture detector (ECD). Fluxes were calculated from the linear increase in gas concentrations over the sampling period. To obtain cumulative sums, fluxes were linearly interpolated and converted to CO₂ equivalent (CO₂ eq) using the global warming potential (GWP, over a 100-year timeframe) of 25 and 298 for CH₄ and N₂O, respectively (IPCC, 2007). The mean of the two measured years was combined with the 4-year mean C balance (i.e. NEP minus DOC export) in CO₂ eq to obtain an estimate of the total GHG balance. Exchanges of CH₄, N₂O, and DOC were not estimated during the snow covered months (January to March) which might have led to underestimation of the total annual exchange. However, winter season fluxes of CH₄, N₂O, and DOC in forests are generally small (Ågren et al., 2007; van Bochove et al., 2000; Yashiro et al., 2006) and thus were assumed to have a negligible effect on the total C and GHG balances in our study.

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2.5 Site quality effects on cumulative NEP

To estimate the total C sequestration from the cumulative NEP over the entire age-sequence, measured annual NEPs from the four sites were linearly interpolated to obtain a 70-year record of annual NEP. Following Peichl et al. (2010a), two scenarios were developed in which NEP of four sites was normalized by differences in their site indices (SI). The SI is a common measure to describe the site quality and is used as predictor for tree growth in forest stands (e.g. Milner, 1992). In the first “low productive” scenario, it was assumed that all four stands grow on low quality sites by applying a SI correction on NEP for TP02 and TP89 to match the SI of TP74 and TP39. Conversely, in the second “high productive” scenario, NEP at TP74 and TP39 was normalized to match the SI of TP02 and TP89 following the assumption that all four stands grow on high quality sites.

3 Results

3.1 Forest C pools and fluxes

Needle litter decomposition rates did not differ significantly among the three older sites with the exponential decay coefficient k ranging from 0.28 to 0.31 (Fig. 1). The rates of needle litter-fall exceeded those of decomposition resulting in a net accumulation of 3.7, 2.0 and 3.2 t C ha⁻¹ yr⁻¹ at the 20, 35 and 70 year-old stands, respectively (Fig. 2).

The largest age-related differences in C pools among sites occurred in stem and root biomass, whereas the canopy (foliage and branches) C pool was similar among the three older stands (Fig. 3). The sum of the secondary C pools, such as woody debris, understory and ground vegetation, were small and similar among three younger sites (ranging from 1.5 to 2.7 t C ha⁻¹) compared to that of the 70 year-old stand (11.3 t C ha⁻¹).

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and 35 year-old) compared to the young and mature stand, respectively. An additional effect of site quality may explain the large difference in NEP and NPP between the two middle-age stands, with considerably higher values noted at the 20 year-old stand characterized by a higher (i.e. double) SI value. In contrast to NEP and NPP, mean GPP and RE increased from the 7 to the 20 year-old stand as well as from the 35 to 70 year-old stand. This indicates that GPP and RE have not yet reached their maxima in this 70 year-old age sequence. Higher GPP and RE at the 20 year-old site, compared to the two older sites, were likely due to the additional effect of higher site quality due to greater water availability at this site (Peichl et al., 2010a). These results based on the mean values for NEP, GPP and RE from biometric and EC estimates agree with those from using estimates from either one of these two methods only since their age-related patterns among sites are generally similar although their absolute values differ between 9 and 56%. A detailed discussion on the underlying reasons for the deviation between the two methods was previously provided by Peichl et al. (2010c). Similar age patterns for forest production and respiration were reported for other forest age-sequences (Goulden et al., 2011; Law et al., 2003; Luysaert et al., 2007; Pregitzer and Euskirchen, 2004). However, in contrast to naturally regenerating forests, the switch from source to sink and the peak in NEP and NPP occurred about 10–20 years earlier in our afforested plantation stands, which highlights their potential for rapid C sequestration.

The contribution of DOC to NEP was considerable at the youngest site (10%), while it was rather low (< 2%) at the three older sites. In agreement with our study, Kindler et al. (2011) found that DOC leaching hardly affected NEP at the majority of the forests investigated in their study. A decrease in soil solution leaching due to greater canopy interception and root water uptake and/or the increase in adsorption of DOC to soil particles due to accumulation of Fe and Al with stand age might explain the reduced DOC export in older forest stands (Camino-Serrano et al., 2014; Kothawala et al., 2009; Peichl et al., 2007). In contrast, a higher contribution of DOC of 11% was observed in a 80 year-old Belgium Scots pine stand (Gielen et al., 2011). Thus, depending on site

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productivity and hydrology, the relative contribution of the DOC export to the forest C balance might vary across forest ecosystems and become significant also at some of the mature forest sites.

The accumulated C sequestration simulated over the entire 70 years of the age-sequence was substantial in both the high and low productive scenarios. Moreover, the difference (by a factor of 2) due to site quality was considerable. The importance of site quality effects on forest growth is widely recognized in traditional forest research (McLeod and Running, 1988; Milner, 1992; Pietrzykowski, 2014; Vose and Allen, 1988). Our study further highlights the need for accounting for differences in site quality when assessing forest C and GHG balances across forest ecosystems and to improve their up-scaling beyond ecosystem boundaries.

4.2 Forest GHG balance across stand age and site productivity

To date, few attempts have been made to quantify the total forest GHG balance and the relative contribution of individual components by including all relevant fluxes (i.e. CO₂, CH₄, N₂O, and DOC) (Ball et al., 2007; Luyssaert et al., 2012; Schulze et al., 2009; Zona et al., 2013). Previous studies estimated GHG balances for individual forests at about +3 tCO₂ eq ha⁻¹ yr⁻¹ for a young short-rotation poplar plantation (Zona et al., 2013) and approximately -15 tCO₂ eq ha⁻¹ yr⁻¹ for a middle-age pine forest (Ball et al., 2007), respectively, which is close to the range observed in our study. Both of these studies also agree with our findings of non-CO₂ GHGs contributing considerably to the GHG balance in recently established plantations, whereas the CO₂ exchange dominates the GHG balance in maturing forests. Similarly, Inatomi et al. (2010), based on results from a process-based model, found that the GHG balance was mainly driven by the CO₂ exchange in a 50 year-old cool-temperate deciduous forest. However, the contribution of non-CO₂ GHG could be substantial in poorly-drained locations within temperate forests (Ullah and Moore, 2011). The manifold variation among the four stands in our study emphasizes that the forest GHG balance may vary widely within

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a heterogeneous (i.e. in terms of age and site quality) forest landscape, which needs to be considered when extrapolating findings from individual forest stands to regions.

Overall, our study advances the current understanding of the forest GHG balance by demonstrating that the magnitude and contribution of individual GHGs may have manifold variations in forests due to differences in stand age and site quality. Both factors determine the forest NEP and thus the magnitude of the CO₂ flux while in comparison having a relatively small effect on the CH₄, N₂O and DOC fluxes. Thus, our findings suggest a link between NEP and the relative contribution of individual gases to the GHG balance with the implication that the ecosystem exchange of CO₂ is the dominant driver of the forest GHG balance in highly productive (e.g. middle-age) forests whereas the contribution of CH₄, N₂O, and DOC fluxes may be more important in low productive (e.g. recently established and mature) forests. A good understanding of the changes in the contribution of individual forest C and GHG fluxes to their total budgets throughout the forest life cycle is imperative when evaluating the potential of these ecosystems as a tool in mitigating global warming and the increase of atmospheric GHG concentrations.

5 Conclusions

We combined C pools and fluxes of CO₂, CH₄, N₂O and DOC to estimate C and GHG balances for an age-sequence (spanning 7 to 70 years) of afforested white pine stands in the temperate region of southern Ontario, Canada. Based on our findings we conclude that:

- The magnitudes and within-stand partitioning of CO₂ fluxes were highly variable with stand age and site quality.
- The role of DOC export for the C balance was substantial in the recently established pine plantation, but low to marginal in the highly productive maturing stands.

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- The combined exchanges of CH₄ and N₂O significantly contributed to the GHG balance of the young and mature pine forests.
- Stand age and site quality may cause manifold differences in the forest C and GHG balances and need to be accounted for to improve the up-scaling from ecosystems to regions.
- Overall, these temperate pine afforestation stands act as a substantial C sink and provide a considerable negative GWP (i.e. cooling effect) throughout the maturing phase.

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Table 1. Site characteristics of the four Turkey Point pine forest stands (adopted from Peichl et al., 2010a).

Site code	TP02	TP89	TP74	TP39
Planting year	2002	1989	1974	1939
Age in 2009 (years)	7	20	35	70
Tree height (m) ^a	3.6 ± 0.4	13.4 ± 0.9	13.5 ± 0.7	22.5 ± 1.4
Tree diameter at breast height (cm) ^a	5.3 ± 0.9	18.2 ± 0.6	16.7 ± 0.3	35.9 ± 5.9
Stem density (stems ha ⁻¹) ^a	1683 ± 189	1317 ± 251	1608 ± 138	421 ± 166
Leaf Area Index (m ² m ⁻²) ^b	1.5	12.8	5.6	8.0
Site index (SI ₂₅) ^c	60	55	28	26
Forest floor C : N ratio	N.A.	16.1	24.5	17.4
Mineral soil C : N ratio (0–10 cm)	11.4	14.2	19.4	15.4
Soil N (0–10 cm) (g m ⁻²)	86	94	67	67
Soil C (0–55 cm) (g m ⁻²)	3720	3390	3670	3670
Soil pH _(CaCl) (0–10 cm)	6.3	4.3	3.7	4.1
Mean water table depth (m)	2–3.5	0.5–2	6–7	6–7

^a In 2008.

^b Chen et al. (2006).

N.A. = not available.

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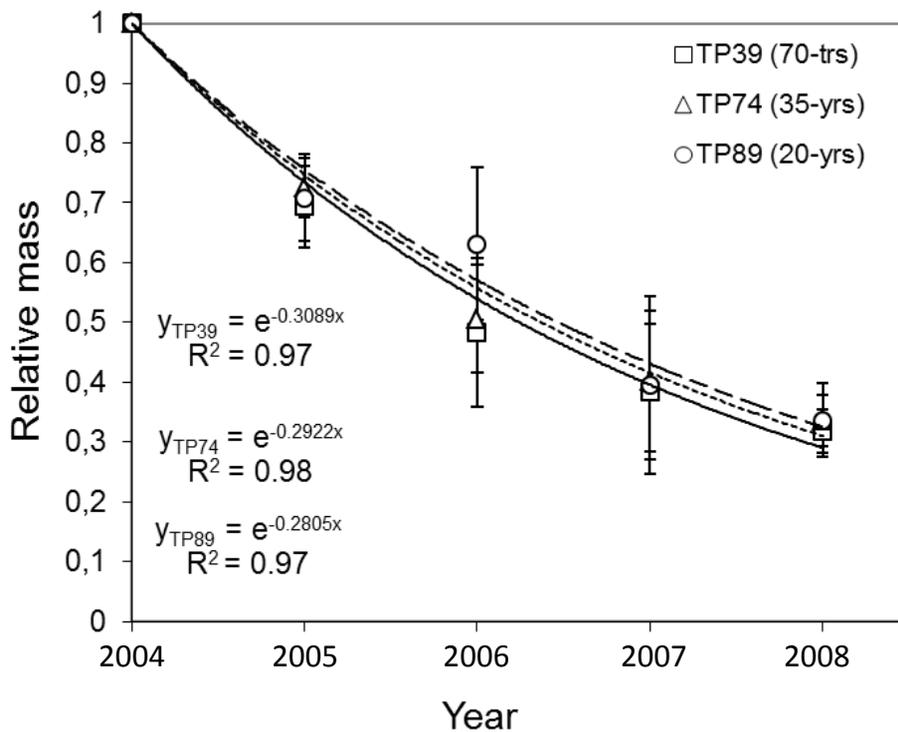


Figure 1. Needle litter decomposition rates over four years (2004–2008) at the three older Turkey Point pine forest sites TP89, TP74 and TP39 (20, 35 and 70 years old, respectively).

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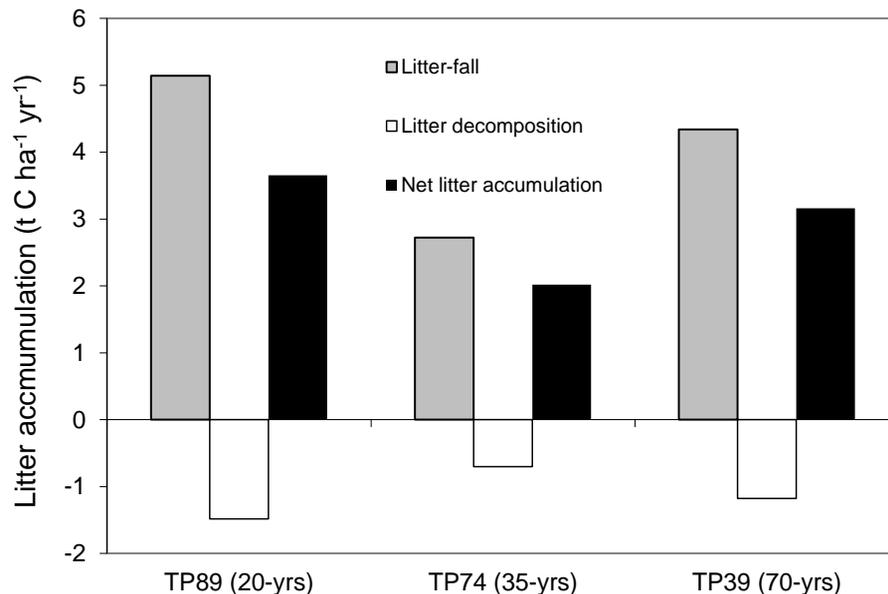


Figure 2. Needle litter-fall, decomposition and net accumulation on the forest floor over four years (2004–2008) at the three older Turkey Point pine forest sites TP89, TP74 and TP39.

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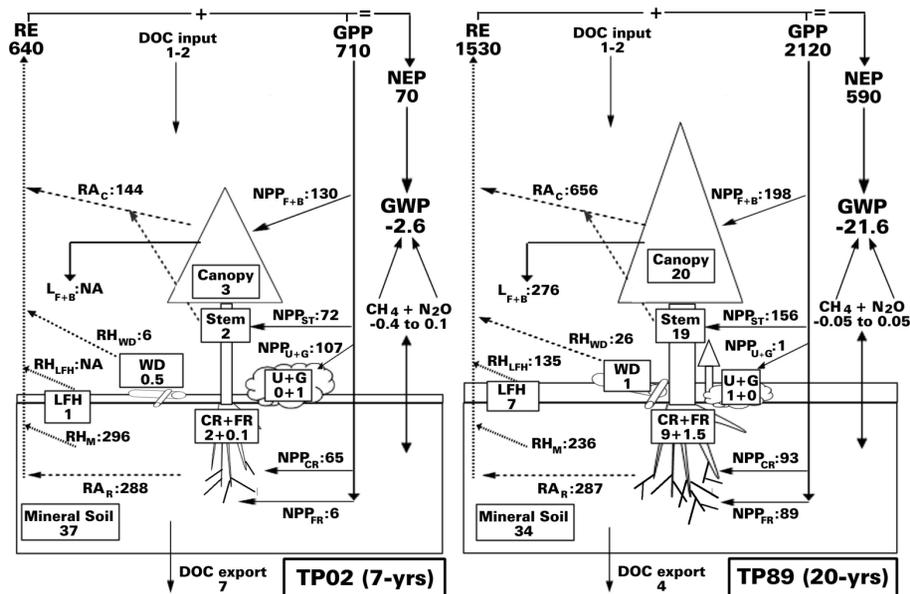


Figure 3a. Ecosystem carbon (C) pools (square boxes; t C ha⁻¹), C fluxes (gross and net primary production (GPP, NPP), litter-fall and DOC flux = solid arrows, ecosystem respiration (RE) fluxes = dotted arrows; g C m⁻² yr⁻¹), CH₄ and N₂O fluxes (t CO₂ eq ha⁻¹ yr⁻¹), and GHG balance expressed as global warming potential (GWP; t CO₂ eq ha⁻¹ yr⁻¹) at the four Turkey Point pine forest sites, TP02, TP89, TP74 and TP39.

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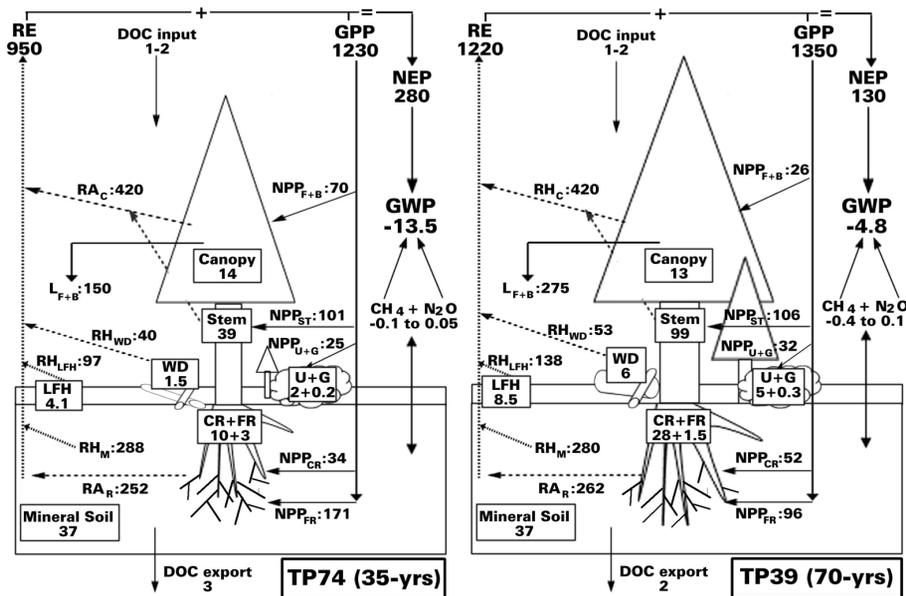


Figure 3b. Continued. Abbreviations: C = canopy; F = foliage; B = branches; ST = stem; LFH = forest floor; WD = woody debris; CR = coarse roots; FR = fine roots; U = understory; G = ground vegetation; M = mineral soil; R = roots; RH = heterotrophic respiration; RA = autotrophic respiration; L = litter-fall, DOC = dissolved organic carbon.

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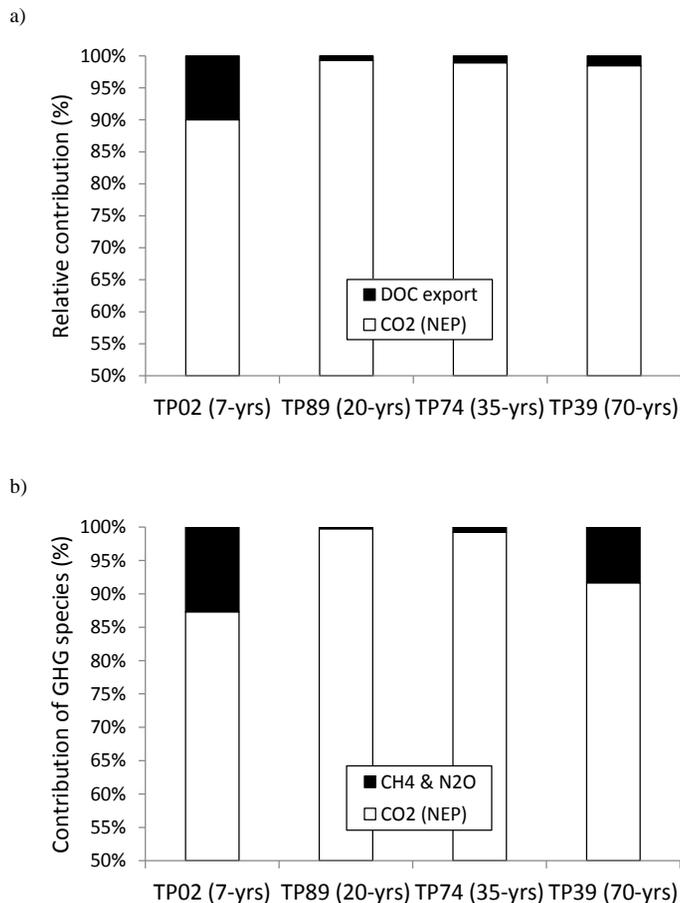


Figure 4. (a) Relative contributions of the DOC export (below 1 m depth) and NEP to the total C balance and (b) relative contribution of the CO₂ exchange and the combined CH₄ and N₂O exchange to the GHG balance at the four Turkey Point pine forest sites, TP02, TP74, TP89, and TP39.

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