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Temporal variability in the sources and fluxes of CO₂ in a residential area in an evergreen subtropical city

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Abstract

Measurements of CO₂ fluxes in temperate climates have shown that urban areas are a net source of CO₂ and that photosynthetic CO₂ uptake is generally not sufficient to offset local CO₂ emissions. However, little is known about the role of vegetation in cities where biogenic CO₂ uptake is not limited to a 2 – 8 months growing season. This study used the eddy covariance technique to quantify the atmospheric CO₂ fluxes over a period of 12 months in a residential area in subtropical Auckland, New Zealand, where the vegetation cover (surface cover fraction: 47%) is dominated by evergreen vegetation. Radiocarbon isotope measurements of CO₂ were conducted at three different times of the day (06:00 – 09:00, 12:00 – 15:00, 01:00 – 04:00) for four consecutive weekdays in summer and winter to differentiate anthropogenic sources of CO₂ (fossil fuel combustion) from biogenic sources (ecosystem respiration, combustion of biofuel/biomass). The results reveal previously unreported patterns for CO₂ fluxes, with no seasonal variability and negative (net uptake) CO₂ midday fluxes throughout the year, demonstrating photosynthetic uptake by the evergreen vegetation all year-round. The winter radiocarbon measurements showed that 85% of the CO₂ during the morning rush hour was attributed to fossil fuel emissions, when wind was from residential areas. However, for all other time periods radiocarbon measurements showed that fossil fuel combustion was not a large source of CO₂, suggesting that biogenic processes likely dominate CO₂ fluxes at this residential site. Overall, our findings highlight the importance of vegetation in residential areas to mitigate local CO₂ emissions, particularly in cities with a climate that allows evergreen vegetation to maintain high photosynthetic rates over winter. As urban areas grow, urban planners need to consider the role of urban greenspace to mitigate urban CO₂ emissions.

Keywords: Biogenic CO₂; Carbon dioxide fluxes; Eddy covariance; Radiocarbon; Source partitioning; Vegetation

1 Introduction

Given the rapid growth of urban areas and increasing per capita emission of anthropogenic carbon dioxide (CO₂), an increasing emphasis is being placed on the design and establishment of low-carbon urban planning and development (Bulkeley, 2013; Rosenzweig et al., 2010). Efforts to establish carbon sinks through urban greening programmes (e.g. planting trees, creating new parkland) are becoming increasingly important in urban planning (Bulkeley, 2013; Pincetl et al., 2012). However, there is still little scientific evidence about the potential of urban greening programmes to mitigate local CO₂ emissions (Weissert et al., 2014). To effectively mitigate urban CO₂ emissions using such greening measures, a good understanding of biogenic CO₂ sources (ecosystem respiration)
and sinks (photosynthetic CO$_2$ uptake) and their comparison with anthropogenic sources (emissions from the combustion of fossil and biofuels/biomass) is fundamental (Pataki et al., 2011).

Traditionally, urban CO$_2$ emissions from fossil fuel combustion have been quantified using bottom-up inventory approaches based on fossil fuel consumption data or energy statistics (Hutyra et al., 2014).

Given that emission proxies (e.g. traffic volume, commercial and residential heating, electricity consumption) are available at a high spatial and temporal resolution, CO$_2$ emissions may be estimated at 1-km or 1-hour resolution. However, high resolution bottom-up inventories are generally limited to local neighbourhoods within a city (e.g. Christen et al., 2011; Velasco et al., 2014) and high resolution inventories for whole cities are still rare (Gurney et al., 2012). Also, bottom-up approaches often only consider anthropogenic CO$_2$ emissions and no information about biogenic CO$_2$ emissions or uptake is included.

In contrast, top-down measurement approaches estimate CO$_2$ fluxes (= net exchanges between the surface and the atmosphere) from measurements of atmospheric CO$_2$ mixing ratios and atmospheric transport models upwind and downwind of cities (e.g. Bréon et al., 2015; Mays et al., 2009; Turnbull et al., 2011b). These top-down approaches are still comparatively new in urban areas. Nonetheless, several studies, mainly in North America and Europe (e.g. Indianapolis (Lauvaux et al., 2016; Mays et al., 2009), Sacramento (Turnbull et al., 2011a), Paris (Bréon et al., 2015) or London (Font et al., 2013)), have used a network of tower and airborne CO$_2$ mixing ratio measurements to estimate urban CO$_2$ fluxes. In addition, recent research has shown that satellite CO$_2$ observations (e.g. GOSAT or OCO2) can be used to quantify urban CO$_2$ emissions (Kort et al., 2012).

These top-down studies have the potential to constrain whole-city CO$_2$ emissions, but provide only limited information about the underlying processes.

The only method to directly measure net CO$_2$ fluxes at the local scale (10$^2$ - 10$^4$ m) is the eddy covariance (EC) method (Velasco and Roth, 2010). The EC method allows a more detailed understanding of processes controlling CO$_2$ fluxes representative of an urban or suburban neighbourhood and provides an opportunity to validate high resolution bottom-up inventories at local scales. However, due to the challenges associated with CO$_2$ flux measurements in heterogeneous environments with large surface roughness parameters and unknown CO$_2$ storage fluxes, EC measurements in urban areas are still limited (Crawford and Christen, 2014; Hutyra et al., 2014; Velasco and Roth, 2010). Most EC studies have been carried out in mid-latitude cities with a focus on CO$_2$ fluxes in urban centres (Grimmond et al., 2004; Helfter et al., 2011; Lietzke et al., 2015; Matses et al., 2009; Nemitz et al., 2002; Velasco et al., 2009), which can be classified as ‘compact low- to high-rise’ or ‘open high-rise’ according to the local climate classification developed by Stewart and Oke (2012). Some studies have measured CO$_2$ fluxes in suburban areas (LCZ, ‘open low-rise’) (Bergeron and Strachan, 2011; Christen et al.,
2011; Coutts et al., 2007; Crawford et al., 2011; Ward et al., 2015). Results from cities with a subtropical or tropical climate where evergreen vegetation takes up CO$_2$ throughout the year are limited to EC measurements in Mexico City (LCZ, compact midrise) (Velasco et al., 2014; Velasco et al., 2005; Velasco et al., 2009) and Singapore (LCZ, compact low-rise) (Velasco et al., 2013). No studies have been undertaken in less populated residential areas (LCZ, open low-rise) in cities with a subtropical climate.

A major challenge in relating both atmospheric CO$_2$ mixing ratio and EC flux measurements to emissions inventories and carbon mitigation techniques is the attribution of CO$_2$ emissions to different CO$_2$ sources. Isotopic analysis, including stable carbon ($\delta^{13}$C) and oxygen ($\delta^{18}$O) isotopes or radiocarbon ($^{14}$C) in CO$_2$, can be used to partition CO$_2$ sources (e.g. Djuricin et al., 2010; Pataki et al., 2003; Turnbull et al., 2006; Wada et al., 2011). Stable carbon isotope data is useful to identify CO$_2$ emissions related to natural gas combustion, since natural gas is more depleted in $^{13}$C than other CO$_2$ sources. However, due to the similar carbon isotope ratios of gasoline/diesel and biogenic respiration, additional tracers are required to identify the contribution of biogenic CO$_2$ sources. Also, sampling is limited to nighttime due to photosynthetic fractionation during daytime.

Preliminary analysis has shown that the application of $\delta^{13}$C in an environment like Auckland, where a) C$_3$ and C$_4$ plants cover a large range of isotopic end-members (-15 to -34‰) similar to natural gas and gasoline isotopic end-members, and b) the majority of households use electric heaters for home heating (61%) (Auckland Council, 2014) is of limited value to partition CO$_2$ sources (see supplementary data).

Some studies have used $\delta^{18}$O (Pataki et al., 2003; Pataki et al., 2007; Wada et al., 2011; Zimnoch et al., 2004) because biogenic respiration has a distinct $\delta^{18}$O signature (Djuricin et al., 2010; Pataki et al., 2007). However, using $\delta^{18}$O to partition urban CO$_2$ sources requires knowledge of the isotopic signature of biogenic respiration, which varies between above and belowground respiration (Pataki et al., 2007). Another tracer to quantify fossil fuel added CO$_2$ in the atmosphere is $^{14}$C since fossil fuel derived CO$_2$ is free of $^{14}$C (Levin et al., 2003; Suess, 1955; Turnbull et al., 2009). Conversely, all other CO$_2$ sources have $^{14}$C:C ratios similar to current atmospheric CO$_2$. Using differences between the observed and background CO$_2$ mixing ratios, $^{14}$C therefore allows to differentiate CO$_2$ added by fossil fuel and CO$_2$ added or removed through biogenic processes (Djuricin et al., 2010; Graven et al., 2009; Meijer et al., 1996; Takahashi et al., 2002; Turnbull et al., 2006).

Overall, currently available studies have shown that urban centres are a net source of CO$_2$, mostly due to high CO$_2$ emissions related to vehicular traffic, home heating and industrial and commercial activities, which exceed photosynthetic CO$_2$ uptake (e.g. Grimmond et al., 2004; Helfter et al., 2011; Lietzke et al., 2015; Matese et al., 2009; Mays et al., 2009; Nemitz et al., 2002; Velasco et al., 2009). Similarly, suburban areas are generally a net CO$_2$ source, however, the influence of biogenic CO$_2$ uptake becomes evident during summer when net CO$_2$
fluxes are occasionally negative (net uptake). In a suburban area in Montreal, Canada, for example, biogenic CO₂ uptake was sufficient to offset summertime CO₂ emissions (Bergeron and Strachan, 2011). Stable and radioisotope data collected at different spatial scales and times of the day show a significant contribution of biogenic respiration ranging from 8.7% during the morning rush hour in central Paris (Lopez et al., 2013) to almost 70% in an urbanised area in Los Angeles at nighttime (Djuricin et al., 2010). These results illustrate that CO₂ from biogenic respiration can also be a dominant source of CO₂.

So far, CO₂ flux studies have generally focused on urban centres and cities in temperate climates and a considerable knowledge gap remains in understanding the CO₂ fluxes and the contribution of biogenic CO₂ sources and sinks in residential urban areas (LCZ, open low-rise), which often cover large areas of cities and where future development due to urban growth will most likely take place. In order to characterize and compare urban CO₂ fluxes and to establish reliable city-wide CO₂ budgets, observations from many sites with contrasting surface cover are essential (Grimmond et al., 2002; Velasco and Roth, 2010). Further, there is a need to combine the EC technique with isotopic tracer measurements to better understand urban CO₂ fluxes and the contribution of anthropogenic and biogenic sources and sinks (Grimmond et al., 2010). Despite the high priority of combining these techniques, no research has yet been published of a combined approach. Moreover, stable and radioisotope analysis in an urban area to identify CO₂ sources is new to cities in the southern hemisphere and generally has only been undertaken in a few other urban areas.

The objectives of this study were to 1) investigate temporal and spatial patterns of residential CO₂ fluxes using EC measurements in a subtropical evergreen dominated low density residential area and, 2) determine processes controlling the diurnal patterns of CO₂ fluxes using CO₂ mixing ratio and δ¹³CO₂ measurements. The results of this study add valuable information regarding the potential of relations between CO₂ flux measurements and δ¹³CO₂ data which enable improved understanding of CO₂ sources in urban areas.

2 Methods

2.1 Study site

The study was conducted from the beginning of June 2014 to the end of May 2015 in the suburb of Botany Downs approximately 15 km south-east of the Auckland city centre (Fig. 1). Auckland is surrounded by the Tasman Sea to the west and the Pacific Ocean to the east and is isolated from downwind anthropogenic pollutants. Auckland’s climate is subtropical and oceanic with mild winters and warm, humid summers (Mackintosh, 2001). Mean local daytime temperatures ranged from 13°C in winter to 20°C in summer throughout the study period. Total annual rainfall (30 year mean) monitored at a nearby (approx. 5 km west of
the study site) weather station is 1212 mm with a minimum in summer and a maximum in winter (Chappell, 2014). The annual rainfall was near normal in 2014 but slightly below average in 2015 (NIWA, 2016).

The area within the EC footprint is representative for a recently developed (< 20 years) residential area in Auckland, which was previously pastureland. One- to two-storey detached dwellings with an average height of approx. 5 m dominate the residential area (height was estimated from 50 dwellings using GoogleEarth). The EC site is on the sportsfield of a primary school about 2 km north of the Botany Downs town centre (36°55'09.3"S 174°55'25.9"E). The EC site has a relatively flat topography (approx. 30 m elevation change within 500 m of the EC site). There is a considerable amount of vegetation (surface cover fraction: approx. 47%) within a 500 m radius of the EC site consisting of residential gardens, a school sportsfield, roadside vegetation and a local parkland with scattered trees to the north (Fig. 1). At around 700 m to the south-east of the EC site the residential area transitions into the rural area, which is dominated by pasture (Fig. 1). Tree species, which account for approx. 17% of the vegetation surface cover fraction, are mostly planted evergreen with few deciduous trees of similar height to the surrounding dwellings, resulting in a similar green cover throughout the whole year. A busy road (23,574 vehicles per weekday (Auckland Transport, 2015)) is located 200 m south / south-east of the tower (Chapel Road). A smaller road (Orangewood Dr) with a weekday average of 3001 vehicles per day transects the EC footprint approx. 120 m north of the tower (Table 1/ Fig. 1). The EC site was exposed to wind from all directions, although south-westerlies and northerlies dominated throughout most seasons (Fig. 2). A Kormann and Meixner (2001) footprint model was used to map the probable flux source area. The model was chosen due to the successful application in other urban areas (e.g. Christen et al., 2011; Lietzke et al., 2015; Liu and Li, 2012). The 50%, 70% and 90% flux source area was on average within 200 m, 500 m and 1300 m of the measurement mast, respectively, but it varied across seasons and time of the day (Fig. 1). At nighttime, when conditions are more stable, the source area can extend several hundreds of meters to the south and south-east (Vesala et al., 2008) (Fig. 1/2). The footprint represented in Fig. 1 is only an indication of the potential source area as to date no footprint model for urban environments effectively considers topography (Helfter et al. 2011).
Table 1. Traffic volume observed at the two main roads within the measurement footprint (Auckland Transport, 2015).

<table>
<thead>
<tr>
<th></th>
<th>Chapel Road</th>
<th>Orangewood Drive</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average daily traffic (Mon – Fri)</td>
<td>23,574</td>
<td>3001</td>
</tr>
<tr>
<td>Average daily weekend traffic</td>
<td>20,428</td>
<td>2220</td>
</tr>
<tr>
<td>Morning peak volume (peak hour)</td>
<td>2570 (07:45)</td>
<td>407 (07:45)</td>
</tr>
<tr>
<td>Midday peak volume (peak hour)</td>
<td>1215 (12:15)</td>
<td>253 (14:00)</td>
</tr>
<tr>
<td>Afternoon/evening peak volume (peak hour)</td>
<td>2282 (17:00)</td>
<td>297 (14:45)</td>
</tr>
</tbody>
</table>

Figure 1. Aerial photograph (LINZ - Land Information New Zealand, 2013) of the residential area overlaid with the flux source area derived from the Kormann and Meixner (2001) source area model for a) daylight vs. nighttime and b) summer vs. winter, c) shows a photograph of the land use and land cover south-east of the tower during winter and d) illustrates the vegetation cover fraction within a 500 m radius of the measurement tower derived from supervised classification using ArcGIS (v.10.2.2).
Figure 2. Wind roses for different seasons and time of the day. The plots show the proportion of time that the wind is from a certain angle and wind speed range. Daylight and nighttime hours were determined using the R package openair (Carslaw & Ropkins, 2015).

2.2 Instrument setup

2.2.1 Eddy covariance CO₂ flux measurements

The eddy covariance instruments were mounted on a pneumatic mast (Hilomast LLC, Debary, US) at 11 m. This height ensured that measurements were taken within the inertial sublayer (see section 2.3.1) (> 2 x roughness element height, (Grimmond et al., 2002)), while also maintaining stability of the mast. The mast was retracted when sustained high winds (> 60 km h⁻¹) and wind gusts (> 80 km h⁻¹) were forecast. A sonic anemometer (Gill WindMaster, Gill Instruments Ltd, Hampshire, UK) was mounted on top of the mast to measure three-dimensional (3-D) wind components and virtual temperature. High speed (10 Hz) CO₂ and water vapour measurements were made with an enclosed infrared gas analyser (LI-7200, LI-COR Inc., Lincoln, NE, USA). Instruments were checked and cleaned every three weeks and the infrared gas analyser was calibrated twice a year.

2.2.2 CO₂ mixing ratio measurements

CO₂ mixing ratios were additionally recorded at approx. 1.2 s intervals using a Picarro isotopic CO₂ gas analyser (G2131-i, Picarro Inc., Santa Clara, CA, US) with the intake at the same height as the eddy covariance system. The Picarro carbon isotope analyser was calibrated weekly using two standard gases (standard 1, CO₂ = 392.82 ppm, δ¹³C = -33.18 ‰; standard 2: CO₂ = 450.50 ppm, δ¹³C = -6.80 ‰). While it is generally recommended to
calibrate the Picarro isotope analyser at least every 10 h (Vogel et al., 2013), this was logistically not feasible. The weekly CO$_2$ drift was on average 1 ppm, which provided an appropriate level of accuracy for our purposes (uncertainty of radiocarbon isotope measurements is around 1 ppm, Table 2). Supporting meteorological data (air temperature, precipitation) was obtained from an air quality monitoring station approx. 2 km north-east of the site (Auckland Council, air quality monitoring programme, unpublished data).

2.2.3 Radiocarbon isotope measurements

To measure the radiocarbon content of CO$_2$ ($^{14}$CO$_2$) we installed three samplers at the study site. The samplers are described in detail by Turnbull et al. (in preparation). Briefly, air was drawn continuously through a synflex inlet line from an intake at 10 m on the measurement mast and passed through an initially carbon-free 1M sodium hydroxide (NaOH) solution to absorb CO$_2$ from the air. For each sample, CO$_2$ was aggregated over three hours on four consecutive days. For example, CO$_2$ was collected during 01:00 - 04:00 for each of four consecutive days, for a total of 12 hours of sampling time for a single sample. Separate samples were taken at three different times of the day 06:00 - 09:00, 12:00 - 15:00 and 01:00 - 04:00 NZST, each for four consecutive weekdays (Tuesday - Friday). These days were chosen based on preliminary analysis, which showed that Mondays often showed a slightly different CO$_2$ mixing ratios pattern compared to other weekdays. The sampling was conducted in summer (Week 1 = 10 – 13 Feb 2015 at 06:00 – 09:00 and 12:00 – 15:00) and winter (Week 2 = 23 – 25 June 2015 at 01:00 – 04:00; Week 3 = 13 – 16 July 2015 at 12:00 – 15:00; Week 4 = 20 – 23 July 2015 at 06:00 – 09:00) (Table 2). Due to a malfunctioning sampler no nighttime data is available for summer. The sampling weeks were chosen to be representative of the general weather pattern for a given season and to avoid public and school holidays.

The timing of the sampling was selected to investigate the contribution of biogenic and anthropogenic CO$_2$ sources at night, during the typically observed morning peak and the afternoon low. In the laboratory, CO$_2$ was extracted from the NaOH by acidification (Turnbull et al., in preparation), reduced to graphite (Turnbull et al., 2015) and the $^{14}$C content was measured using accelerator mass spectrometry at GNS Science, New Zealand (Zondervan et al., 2015). A small amount of carbon is always present in the prepared NaOH solution and a blank correction was made to account for this, based on the CO$_2$ content of an aliquot of the same NaOH solution batch that was not exposed to air.

2.2.4 Soil CO$_2$ efflux
Using a CO$_2$ infrared gas analyser (EGM-4, PP-Systems Inc., Amesbury, MA, USA) and a soil respiration chamber SRC-1 (diameter: 10 cm, height: 25 cm) soil CO$_2$ efflux was measured for a winter and summer night (1 hour after sunset) at the school sportsground surrounding the EC site and the parkland 200 m north of the EC site. Soil CO$_2$ efflux was measured at nine points within each site over a period of 100 s. These measurements were used to estimate the CO$_2$ added from heterotrophic respiration (section 2.4). The sportfield and parkland are regularly mowed (clippings left on the ground), but not irrigated. The sportfield is also fertilized three times a year.

2.3 Data processing and data quality

2.3.1 Eddy covariance CO$_2$ flux measurements

CO$_2$ flux data from 1 June 2014 to 31 May 2015 were analysed. Raw data were processed at 30 min intervals. Turbulent fluxes were calculated as the mean covariance between the instantaneous deviations of the vertical wind velocity and mixing ratio (Burba, 2013) using EddyPro Advanced (v.5.1.1, LI-COR) (EQ. 1).

$$F_{CO_2} = \overline{w'} \times \overline{\rho_c'}$$  

EQ. 1

where $F_{CO_2}$ is the CO$_2$ flux, $\overline{w'} \times \overline{\rho_c'}$ is the covariance between the mean instantaneous deviations in vertical wind velocity ($w'$) and the density of CO$_2$ in the air ($\rho_c'$) (Burba, 2013). Positive $F_{CO_2}$ values indicate fluxes towards the atmosphere (net emissions) and fluxes towards the surface (net uptake) are negative. The flux processing followed standard procedures used in other suburban studies (Bergeron and Strachan, 2011; Crawford et al., 2011; Ward et al., 2013). Primary processing included de-spiking of raw data, double coordinate rotation, correction for time lag through covariance maximization and block averaging. A set of corrections (angle of attack correction for wind components (Nakai and Shimoyama, 2012), spectral corrections (Moncrieff et al., 2004; Moncrieff et al., 1997)) was also applied. No Webb-Pearman-Leuning (WPL) (Webb et al., 1980) density corrections are needed for the LI-7200 since it outputs the dry mixing ratio of CO$_2$ (Burba, 2013). In addition, quality control flags (QC) were calculated following Mauder and Foken (2004) using a combination of the steady state and developed turbulent conditions test (QC flags 0 - 2, with 0 being the best quality). Quality control of the processed fluxes resulted in the rejection of low quality data (QC flag = 2) and where raw data were outside absolute limits, showed drop-outs or discontinuities based on statistical tests output by EddyPro.

Negative nighttime CO$_2$ fluxes, which were likely due to low mechanical shear in the stable boundary layer
(Ramamurthy and Pardyjak, 2015), were also removed (1.3%). Similar to other urban studies (e.g. Bergeron and Strachan, 2011; Crawford et al., 2011; Ward et al., 2013) no filtering was used for low friction velocity ($u^* < 0.1 \text{ m s}^{-1}$), however only 6% of the final data had a friction velocity $< 0.1 \text{ m s}^{-1}$. No corrections for CO$_2$ storage were made in this study due to the large uncertainty related to the single level calculation method at the height of the EC instruments (Crawford and Christen, 2014). However, CO$_2$ storage correction estimates using the single level method indicate that the difference in the CO$_2$ flux due to CO$_2$ storage is below 1 µmol m$^{-2}$ s$^{-1}$ across the median diurnal cycle, which was the focus of this study, and the mean CO$_2$ flux difference across all observations is below 1%. Energy balance closure was not assessed in this study due to the difficulty of determining it reliably in urban environments (Bergeron and Strachan, 2011).

To ensure that the eddy covariance system was measuring turbulent fluxes we analysed power spectra and co-spectra (Kaimal and Finnigan, 1994). Following procedures described by Moore et al. (2015) we used normalized ensemble averaged data, which were sorted by time of day (12:00 – 14:00) and averaged for five consecutive days for each month and different wind directions. The power spectra and co-spectra were compared to standard power curves (Kaimal and Finnigan, 1994). The power spectra and co-spectra followed the theoretical $-2/3$ and $-3/4$ slopes in the inertial subrange (see supplementary data), suggesting that the eddy covariance system was capable of measuring turbulent fluxes despite the comparably low measurement height.

After quality control 66% of the CO$_2$ flux ($F_{\text{CO}_2}$) data were available for the data analysis. Gaps were relatively evenly distributed across the year with the exception of November (14/11/2014 - 03/12/2014) when the LI-7200 was calibrated and serviced. Gap-filling procedures for CO$_2$ flux measurements in urban environments vary in the literature and most studies avoid the errors associated with procedures by averaging diurnal fluxes over monthly time periods instead (Menzer et al. 2015). Therefore, given the day to day variability of Auckland’s climate and the lack of a long term (>1 year) CO$_2$ flux dataset for this site, artificial gap-filling was not used in this dataset.

All results are reported in local standard time (NZST) on a 24-hour period, since emission activities (i.e. traffic) occur at local time. Using local standard time includes the daylight saving time and therefore results in a shift in the morning rush hour peak during the summertime. Nighttime and daylight periods were calculated using the R package openair and refer to hours after sunset and before sunrise (Carlslaw and Ropkins, 2012).

2.3.2 CO$_2$ mixing ratios

Similar to the CO$_2$ flux data, CO$_2$ mixing ratio data from 1 June 2014 to 31 May 2015 was used in this study.

Data availability was higher for the CO$_2$ mixing ratio from the isotopic CO$_2$ gas analyser with only 19% of the
data being discarded due to instrument calibration and mast retraction. The mixing ratio data were averaged to
30 min intervals to match the eddy covariance data. The largest gap in mixing ratios occurred in December due
to pump failure. The data post processing and analysis was undertaken using R (v.3.1.0). All results are reported
in local standard time (NZST) on a 24-hour period.

2.3.3 Calculation of fossil fuel CO$_2$ (CO$_2^{\text{ff}}$) from $^{14}$CO$_2$

To quantify CO$_2$ added from fossil fuels from $^{14}$CO$_2$ we used EQ. 2 following equation 3 in Turnbull et al.
(2009),

$$CO_{2}^{\text{ff}} = \frac{CO_{2}\text{obs}(\Delta_{\text{obs}} - \Delta_{\text{bg}})}{\Delta_{\text{ff}} - \Delta_{\text{bg}}} - \beta$$  \hspace{1cm} \text{EQ. 2}

where $CO_{2}\text{obs}$ is the mean CO$_2$ mixing ratio measured at the study site during the sampling period, $\Delta_{\text{obs}}$, $\Delta_{\text{bg}}$ and
$\Delta_{\text{ff}}$ are the $\Delta^{14}$C of the sample, the background and fossil fuel (-1000‰), respectively. Since we do not have well
characterised local background data for $^{14}$C, we used measurements from Baring Head, NZ (Currie et al., 2011);
extended to 2015 (Turnbull, unpublished data) (Table 2). Model results suggest that the Baring Head
measurements could be up to 1‰ lower than in Auckland (Turnbull et al., 2009) potentially introducing a bias in
CO$_2^{\text{ff}}$ of up to 0.3 ppm which we do not account for. The second term of EQ. 2 ($\beta$) represents the correction for
any other CO$_2$ sources and sinks (i.e. heterotrophic respiration) (EQ. 3),

$$\beta = \frac{CO_{2}\text{other} (\Delta_{\text{other}} - \Delta_{\text{bg}})}{\Delta_{\text{ff}} - \Delta_{\text{bg}}}$$  \hspace{1cm} \text{EQ. 3}

Heterotrophic respiration typically dominates this term and previous studies have shown that it is typically 0.2 -
0.5 ppm (Miller et al., 2012; Turnbull et al., 2009; Turnbull et al., 2006). We calculated this explicitly for
Auckland, such that CO$_2$ added by heterotrophic respiration (CO$_2^{\text{other}}$) was estimated using chamber based soil
CO$_2$ efflux measurements in the surroundings of the EC site and assuming that heterotrophic respiration
accounts for approx. 50% of the total soil CO$_2$ efflux (based on an average for managed temperature grassland
and pasture systems (Bond-Lamberty and Thomson, 2014)). We estimated $\Delta_{\text{other}}$ as 174 ± 100‰, determined
from the 27 year mean residence time of surface soil carbon estimated for volcanic soils near Auckland (Baisden
et al., 2011) and the Baring Head atmospheric $\Delta^{14}$C 27 years earlier than our samples were collected. This
resulted in a slightly higher $\beta$ than other studies have used (Table 2), largely due to the long residence time of soil carbon in Auckland’s volcanic soils.

We determine an uncertainty of around 1 ppm for the calculated fossil fuel CO$_2$ based on the $\Delta^{14}$C measurement uncertainties and uncertainty in $\beta$ (Table 2), but this uncertainty does not account for any biases in $\beta$ or in $\Delta$bg.

For each sampling period the mean biogenic CO$_2$ (CO$_2$bio), including soil/leaf respiration, photosynthesis and CO$_2$ added from the combustion of biofuels and biomass, was calculated using EQ. 4 (Graven et al., 2009; Turnbull et al., 2006),

$$\text{CO}_2\text{bio} = \text{CO}_2\text{obs} - \text{CO}_2\text{bg} - \text{CO}_2\text{ff}$$

where CO$_2$obs is mean CO$_2$ mixing ratio measured at the study site during the sampling period, CO$_2$bg the background CO$_2$, which was obtained from measurements at Baring Head (Table 2) (Brailsford et al., 2012; Stephens et al., 2013), and CO$_2$ff the CO$_2$ added from fossil fuels. The main uncertainty in CO$_2$bio is the propagated CO$_2$ff uncertainty (around 1 ppm, Table 2), and we note that any bias in CO$_2$ff will propagate directly into the CO$_2$bio estimate. However, there is large variability in CO$_2$ observed (mostly due to meteorology, e.g. boundary layer height) and the absolute value of CO$_2$bio varies depending on the observed CO$_2$. The proportions (%) are less influenced by changes in meteorology and are therefore reasonable assumptions of the biogenic CO$_2$ source contribution at night, during the commonly observed morning peak and the afternoon low.

2.6 Back trajectory analysis

Back trajectory analysis was undertaken to support the $^{14}$CO$_2$ data. We used pre-calculated back trajectories from the HYPSLIT trajectory model (Hybrid Single Particle Lagrangian Integrated Trajectory Model) based on the global NOAA-NCEP/NCAR reanalysis meteorological data (Stein et al., 2015). The trajectories were calculated at 3-hour intervals starting at 10 m and propagated backwards in time (Carslaw, 2014). The R package openair was used to plot the back trajectories (Carslaw and Ropkins, 2012; Carslaw and Ropkins, 2016).

3 Results and discussion

3.1 Temporal and spatial variability of CO$_2$ fluxes

CO$_2$ fluxes at the Auckland residential study site ranged from -23 $\mu$mol m$^{-2}$ s$^{-1}$ to 37 $\mu$mol m$^{-2}$ s$^{-1}$. Although negative CO$_2$ fluxes were observed across the whole year, the mean (median) CO$_2$ flux across all observations
remained positive (1.77 ± 3.94 μmol m⁻² s⁻¹ (2.32 μmol m⁻² s⁻¹)), indicating that the residential study site was a net source of CO₂ for the study period. Converting the mean CO₂ flux to an annual CO₂ flux would result in approx. 25 t ha⁻¹ yr⁻¹. However, this mean CO₂ may be a slight overestimate due to the larger data gap in November, when CO₂ fluxes are likely to be lower as a result of longer days allowing for longer periods of photosynthetic CO₂ uptake. Aggregating mean daily weekend and weekday CO₂ fluxes (calculated using days when data capture was >75%) for each month resulted in an annual net CO₂ flux of approx. 22 t ha⁻¹, which is slightly lower than the mean annual net CO₂ flux across all observations (approx. 25 t ha⁻¹ yr⁻¹). Both estimates have errors related to uncertainties resulting from missing data and a longer term (>1 year) dataset is required to increase confidence in the net annual CO₂ release of this residential study site.

The mean CO₂ flux at the Auckland residential study site (1.77 μmol m⁻² s⁻¹) is slightly below results from suburban areas with similar vegetation cover, such as Montreal, Canada (3.75 μmol m⁻² s⁻¹, Bergeron and Strachan, 2011), Melbourne, Australia (3.21 μmol m⁻² s⁻¹, Coutts et al., 2007), but comparable to findings from Swindon, UK (1.26 μmol m⁻² s⁻¹, Ward et al., 2015). Differences in climate (e.g. cold vs. mild winter), vegetation type (deciduous vs. evergreen tree species), traffic load and footprint composition (e.g. proximity to local CO₂ sources such as nearby roads) are key drivers of CO₂ fluxes and likely explaining the variability observed across different sites. Compared to non-urban ecosystems in comparable regional and climatic settings, the mean CO₂ flux at the Auckland residential study site is more positive (a larger CO₂ source). For example, the yearly mean CO₂ flux of a managed pasture (Rutledge et al., 2015) and a New Zealand pine forest (Arneth et al., 1988) were -0.12 μmol m⁻² s⁻¹ and -0.52 μmol m⁻² s⁻¹, respectively, demonstrating the potential influence of anthropogenic CO₂ emissions and a lower vegetation cover in the EC footprint.

The diurnal variability of CO₂ fluxes, analysed separately for different seasons, weekdays and weekends is illustrated in Fig. 3. The diurnal CO₂ flux pattern at the Auckland residential site showed little seasonal variability. In fact, negative (net uptake) CO₂ fluxes from 10:00 until almost 18:00 in summer and 14:00 in winter, when days were shorter, was of similar magnitude across the whole year (Fig. 3). Daytime CO₂ uptake was lowest (less negative) when wind flow was from the south and south-west where the vegetation cover was lowest (Fig. 1/5). This illustrates the photosynthetic CO₂ uptake potential of evergreen vegetation during winter. While some studies (e.g. Bergeron and Strachan, 2011; Crawford et al., 2011; Ward et al., 2013) observed negative midday CO₂ fluxes during the growing season, this is the first study that showed CO₂ uptake at midday across all seasons. The few other studies from cities with a subtropical and tropical climate were undertaken in compact mid and low-rise areas with a substantially lower vegetation cover (max. 16%) and fluxes were mostly positive throughout the year (Velasco et al., 2014; Velasco et al., 2005; Velasco et al., 2009; Velasco et al.,...
In Melbourne, where vegetation consisted of evergreen and deciduous trees, photosynthetic CO$_2$ uptake during winter was important too, however, it was smaller than during summer (Coutts et al., 2007). Also, mean net CO$_2$ fluxes remained positive at all hours of the day at the residential site in Melbourne. The EC site in Melbourne was close (< 50 m) to two major roads with an almost three times higher traffic volume than observed in this study, which likely had a large influence on the CO$_2$ fluxes observed in the Melbourne suburban area. In addition, it is possible that photosynthetic CO$_2$ uptake in Melbourne was limited by low water availability due to drought conditions (Coutts et al., 2007).

The median nighttime CO$_2$ fluxes were more positive (a larger CO$_2$ source) than median daytime CO$_2$ fluxes at the residential study site. This is opposite to results from urban and suburban areas where high anthropogenic CO$_2$ emissions during daytime were reported, which resulted in larger daytime CO$_2$ fluxes (e.g. Bergeron and Strachan, 2011; Liu et al., 2012). Unlike typically observed in cities with cold winters where CO$_2$ emissions related to heating contribute considerably to nighttime CO$_2$ fluxes (e.g. Bergeron and Strachan, 2011; Crawford et al., 2011), median nighttime CO$_2$ fluxes were lowest during winter (3.12 µmol m$^{-2}$ s$^{-1}$) (Fig. 3). The majority of households (approx. 60%) in east Auckland use electric heaters for home heating according to a household survey (Auckland Council, 2014). The remaining households use natural gas (approx. 20%) and wood burners (approx. 20%) (Auckland Council, 2014). Thus, nighttime CO$_2$ emissions due to home heating are likely small in this residential area, suggesting that biogenic respiration is probably the major source of CO$_2$ at nighttime. The source footprint at nighttime extended several hundred meters to the east and southeast into the rural area due to more stable conditions, likely resulting in a stronger contribution from biogenic CO$_2$ emissions (Fig. 1). This is also seen in Fig. 4, suggesting higher (more positive) nighttime CO$_2$ fluxes when wind was from the more vegetated area east of the measurement mast. The soil CO$_2$ efflux measurements within the EC footprint agree with the seasonal pattern we observed at our residential site with lower values observed in winter (Jun) (5.8 ± 2.2 µmol m$^{-2}$ s$^{-1}$) than summer (Feb) (6.7 ± 1.7 µmol m$^{-2}$ s$^{-1}$). A similar magnitude and seasonal pattern of soil CO$_2$ efflux was also measured across other urban parkland and forest sites in Auckland (Weissert et al., 2016). However, the magnitude of the chamber measurements is higher than the mean nighttime EC measurements during the chamber sampling measurement period (2.85 ± 0.88 µmol m$^{-2}$ s$^{-1}$ and 2.99 ± 1.13 µmol m$^{-2}$ s$^{-1}$ in Jun and Feb, respectively).

Nighttime CO$_2$ fluxes measured by the eddy covariance are often underestimated during periods of insufficient turbulent mixing and the role of drainage flows of CO$_2$ has been discussed in various studies from non-urban environments (e.g. Aubinet et al., 2008, Baldocchi et al., 2000; Massman and Lee, 2002). Typically, such periods are removed using a friction velocity criterion. However, periods with insufficient turbulence are rarely
observed in urban areas due to higher surface roughness and a more open canopy structure (Crawford et al., 2011) and isolated cases are typically not removed in urban studies due to the variation and uncertainty in estimating a friction velocity ($u^*$) threshold (e.g. Bergeron and Strachan, 2011; Crawford et al., 2011; Ward et al., 2013). In this data set, removing periods when $u^*$ was below 0.1 m s$^{-1}$ increased the nighttime CO$_2$ fluxes by less than 0.5 µmol m$^{-2}$ s$^{-1}$, suggesting that the effects of very calm conditions and drainage flows were negligible. The divergence between the EC and the chamber-based measurements in this study is therefore more likely due to the fact that the EC CO$_2$ flux is representative of a blended average of the source area, which also includes built areas (Järvi et al., 2012). Thus, the few chamber-based soil CO$_2$ efflux measurements are unlikely to be representative of the net CO$_2$ flux measured at the EC flux tower and it would be necessary to measure soil CO$_2$ efflux at a higher spatial and temporal resolution to make any direct comparisons.

The weekday diurnal CO$_2$ flux reached a peak in the morning (08:00) with median CO$_2$ fluxes ranging between 2.96 µmol m$^{-2}$ s$^{-1}$ and 4.50 µmol m$^{-2}$ s$^{-1}$ across different seasons (Fig. 3). Larger positive CO$_2$ fluxes were associated with wind directions from the east and south-west (Fig. 4). This morning peak may partly be attributed to increasing turbulence and the release of CO$_2$ stored during nighttime, which increases the positive CO$_2$ flux (Crawford and Christen, 2014). However, peak CO$_2$ flux at 08:00 coincides with the peak traffic density, which is observed at 07:45 (Table 1). The magnitude of the peak is substantially lower than observed in the residential area in Melbourne (14 µmol m$^{-2}$ s$^{-1}$ – 17 µmol m$^{-2}$ s$^{-1}$, Coutts et al. (2007)), demonstrating the smaller influence from traffic CO$_2$ emissions at our site in the Auckland residential area. A similar morning peak was not observed during the weekend, providing further evidence that the peak is related to anthropogenic CO$_2$ emissions during the morning rush hour.

Averaged across all hours of the day, CO$_2$ fluxes were 28% lower during the weekend than on weekdays. As expected, the weekday-weekend differences were most evident during morning rush hour (08:00) when weekend CO$_2$ fluxes were reduced by just over 40%, due to a lower traffic volume on weekends (Table 1). Weekend CO$_2$ fluxes were similar to those on weekdays for the remainder of the day, however, weekend CO$_2$ fluxes were slightly lower than weekday CO$_2$ fluxes after 14:00 until about 22:00. This implies that the influence of traffic CO$_2$ emissions was low and possibly spread across more hours in the afternoon and evening. This is also visible in the traffic data, which shows that the peak afternoon/evening traffic was at 14:45 (when school finishes) on Orangewood Drive and at 17:00 on Chapel Road (Table 1).
Figure 3. Median diurnal variability of a) CO$_2$ fluxes across different seasons (the shaded area represents the inter-quartile range) and b) weekday vs. weekend CO$_2$ fluxes.

Figure 4. Polar annulus plot of CO$_2$ fluxes as a function of wind direction and time of day for June 2014 - June 2015. The 0 inside of the annulus represents midnight, while the 23 outside of the annulus represents 23:00.
3.2 Temporal and spatial variability of CO₂ mixing ratios

In contrast to CO₂ fluxes, which are primarily driven by emission and uptake of CO₂ in the local surroundings, CO₂ mixing ratios likely have a larger footprint and also depend on meteorological processes, such as boundary layer height or wind speed (Reid and Steyn, 1997; Velasco et al., 2005; Vesala et al., 2008). The relationship between CO₂ mixing ratios and wind speed is shown in Fig. 5. Maximum CO₂ mixing ratios were observed when wind speed was below 2 m s⁻¹, which limits the dispersion of CO₂. The CO₂ mixing ratio generally decreased with increasing wind speed, due to improved advection, reaching a minimum when wind speed was > 6 m s⁻¹ (Fig. 5).

Similar to CO₂ fluxes, CO₂ mixing ratios showed little seasonal variability with median values ranging from 398 ppm during spring to 402 ppm in autumn. The median diurnal pattern of CO₂ mixing ratios is consistent with the CO₂ fluxes at midday when a minimum is reached, but different at night and early morning due to boundary layer growth and collapse, respectively (Fig. 6). CO₂ mixing ratios decreased continuously after 08:00 and remained low (395 – 396 ppm) occasionally reaching values below background at midday (Table 2). CO₂ mixing ratios started rising again about half an hour before sunset (winter: approx. 17:30, summer: approx.: 20:30) when turbulence decreased (Fig. 6). Median nighttime CO₂ mixing ratios were higher in summer than winter, possibly due to higher soil respiration (Fig. 3, section 3.2) and low wind speeds (< 1.9 m s⁻¹) (Fig. 6), which slowed the dispersion of biogenic respired CO₂, possibly resulting in CO₂ storage in the air volume below the measurement height (Crawford and Christen, 2014). However, stable (z’/L (Monin-Obhukov stability parameter) > 0.1) nighttime conditions were generally more common during winter (43%) than summer (33%), which occasionally also led to high nighttime build-up of CO₂ mixing ratios during winter (Fig. 6). The diurnal pattern observed at our site is similar to that observed in other cities (Coutts et al., 2007; Rice and Bostrom, 2011; Velasco et al., 2009; Vogt et al., 2006). The weekday-weekend differences were less pronounced than observed with the CO₂ fluxes. The morning peak is still evident, however, CO₂ mixing ratios were almost identical during the afternoon hours (Fig. 6). This may be due to a deeper boundary layer and stronger vertical mixing typical for midday hours, which dilutes CO₂ emissions (Strong et al., 2011). Interestingly, weekend nighttime CO₂ mixing ratios exceeded those during weekdays, which was not observed from the CO₂ fluxes (Fig. 6). This may be related to the social nighttime activities on weekends that lead to higher traffic outside the CO₂ flux footprint (Velasco et al., 2009).
Figure 5. Hourly averaged wind speed versus CO$_2$ mixing ratios measured between June 2014 and June 2015.

Figure 6. Median diurnal variability of a) summer and winter CO$_2$ mixing ratios (the shaded area represents the inter-quartile range), b) weekday and weekend CO$_2$ mixing ratios and c) the median summer and winter diurnal variability of wind speed.
3.3 CO$_2$ source analysis using $^{14}$CO$_2$

The seasonal and diurnal CO$_2$ flux and mixing ratio patterns indicated some influence of anthropogenic CO$_2$ emissions during the morning rush hour on weekdays, but little anthropogenic CO$_2$ emissions for the remainder of the day (see 3.1 and 3.2). To get a better understanding of the CO$_2$ sources controlling the diurnal patterns of CO$_2$ mixing ratios (and CO$_2$ fluxes) we used $^{14}$CO$_2$ measurements.

$\Delta^{14}$CO$_2$ varied from +30.00 to -10.10‰, with lowest values measured in winter and during the morning rush hour sample, demonstrating a stronger influence from $^{14}$C-free fossil fuel CO$_2$ (Table 2). In fact, fossil fuel CO$_2$ accounted for 85% (CO$_2$ff = 12.71 ± 1.07 ppm) of the CO$_2$ enhancement in the morning (06:00 – 09:00) during the winter sampling period in week 4 (20 – 23 July 2015) when wind was in line with residential and industrial areas (Table 2, Fig. 7).

No fossil fuel CO$_2$ was detected during the morning rush hour sample in summer when wind was predominantly from the south-east (Fig. 1, Table 2). Unlike CO$_2$ fluxes, CO$_2$ and $^{14}$CO$_2$ mixing ratios potentially have a much larger footprint (Vesala et al., 2008). The origin of the air mass for this sampling period was south-east of the study site (Fig. 7). The land cover transitions from residential to rural, dominated by pasture, at around 700 m to the south-east of the study site (Fig. 1), potentially explaining the strong biogenic contribution. This suggests that the seasonal differences observed in these single samples are likely due to differences in wind direction and CO$_2$ source footprint rather than seasonal variability in CO$_2$ sources.

Similarly to the summer morning sample period, we observed a large positive mean biogenic contribution during our single nighttime sample in winter (23 – 25 June 2015) (5.40 ± 0.99 ppm) and no detectable (< 1 ppm) fossil fuel CO$_2$ contribution. This implies all the nighttime CO$_2$ build up was of biogenic origin (Table 2). While the mean CO$_2$bio is likely dominated by soil and plant respiration, it is possible that biomass burning from home heating contributed as well. Approx. 20% use wood burners for home heating in east Auckland (Auckland Council, 2014). Other studies that used carbon isotopes to partition urban CO$_2$ sources also reported high contributions from biogenic CO$_2$ at nighttime, although fossil fuel CO$_2$ was still present. In an urbanised area in Los Angeles, US for example, CO$_2$bio accounted for almost 70% of the nighttime CO$_2$ during spring (Djuricin et al., 2010). In a residential neighbourhood in Salt Lake City, US almost 60% of nighttime CO$_2$ was of biogenic origin during spring (Pataki et al., 2003). However, both studies observed much lower biogenic contributions during winter (approx. 40%), when aboveground respiration was reduced (Djuricin et al., 2010; Pataki et al., 2003). The studies in Los Angeles (Djuricin et al., 2010) and Salt Lake City (Pataki et al., 2003) showed large contributions from natural gas combustion related to domestic heating or cooling. Differences among studies can
therefore largely be explained by differences in climate and footprint composition (i.e. vegetation cover as well as vegetation type).

The midday \( \Delta^{14} \text{CO}_2 \) was close to background during the winter sampling period and slightly above background during the summer sampling week possibly due to \( \text{CO}_2 \) emissions related to heterotrophic respiration, which have a \( \Delta^{14} \text{CO}_2 \) well above background (174\( \% \pm 100\% \)) (Table 2). Wind was predominantly from the east during both sampling periods, with air masses passing over the rural Auckland area prior to arrival at the study site (Table 2, Fig. 7). At midday, mean biogenic \( \text{CO}_2 \) was negative during both sampling weeks and we did not detect any \( \text{CO}_2 \) from the combustion of fossil fuels between 12:00 and 15:00, suggesting that photosynthetic \( \text{CO}_2 \) uptake offset \( \text{CO}_2 \) respired by vegetation and soils at midday during the sampling periods. This agrees with the negative \( \text{CO}_2 \) flux measured at the site between 12:00 and 15:00, which was particularly evident when wind direction was from the more vegetated area to the east (Fig. 1, Table 2).

These results suggest, that the overall contribution from anthropogenic \( \text{CO}_2 \) was low at the residential study site, particularly when wind was from the east / southeast. There was a significant CO\text{ff} contribution during morning rush hour and it is possible that there is also a CO\text{ff} contribution in the (unsampled) evening rush hour period. Such information could not be obtained using \( \text{CO}_2 \) flux data alone demonstrating the value of linking \( \text{CO}_2 \) flux measurements with \( ^{14} \text{CO}_2 \) data. Although the \( ^{14} \text{CO}_2 \) results agreed well with the findings from the EC \( \text{CO}_2 \) flux, it is important to bear in mind the possible differences between the \( \text{CO}_2 \) flux and \( \text{CO}_2 \) (\( ^{14} \text{CO}_2 \)) mixing ratio source footprint. Also, given the very small number of samples, the results need to be interpreted with caution as the findings might not be representative for other times of the year. Similarly, sampling in this study was limited to weekdays and future research may investigate the contribution of anthropogenic and biogenic \( \text{CO}_2 \) sources on weekends, particularly in the evening when the so called ‘party effect’ was observed (Fig. 6).
Table 2. Calculated fossil fuel contribution to CO\textsubscript{2} mixing ratios and data used in EQ. 2 and 3 as well as the mean CO\textsubscript{2} flux, wind speed and wind direction measured during the sampling period with eddy covariance. The parameters refer to EQ. 2 and 3 (\(\Delta_{\text{obs}} = \Delta^{14}\text{C} \) of the sample, \(\Delta_{\text{bg}} = \Delta^{14}\text{C} \) of the background, \(\Delta_{\text{ff}} = \Delta^{14}\text{C} \) of fossil fuel, \(\beta \) = correction for CO\textsubscript{2} added from heterotrophic respiration). A negative CO\textsubscript{2}ff is physically not possible, however, it is within two sigma of zero and therefore negligible.

<table>
<thead>
<tr>
<th></th>
<th>Summer</th>
<th>Winter</th>
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<tr>
<td></td>
<td>Week 1\textsuperscript{11}</td>
<td>Week 2</td>
</tr>
<tr>
<td>06:00 – 09:00</td>
<td>407</td>
<td>395</td>
</tr>
<tr>
<td>12:00 – 15:00</td>
<td>12:00 – 15:00</td>
<td>01:00 – 04:00</td>
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<tr>
<td>Mean CO\textsubscript{2}obs (ppm)</td>
<td>407</td>
<td>395</td>
</tr>
<tr>
<td>(\Delta_{\text{obs}} ) (‰)</td>
<td>30.00</td>
<td>26.50</td>
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<tr>
<td>(\Delta_{\text{bg}} ) (‰)</td>
<td>24</td>
<td>24</td>
</tr>
<tr>
<td>(\Delta_{\text{ff}} ) (‰)</td>
<td>-1000</td>
<td>-1000</td>
</tr>
<tr>
<td>(\beta ) (ppm)</td>
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<td>-1.0</td>
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<tr>
<td>Mean CO\textsubscript{2}bg</td>
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<td>397</td>
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<tr>
<td>CO\textsubscript{2}ff (ppm)</td>
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<tr>
<td>Uncertainty (ppm)</td>
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<td>100/0</td>
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<td>Median CO\textsubscript{2} flux (µmol m\textsuperscript{-2} s\textsuperscript{-1})</td>
<td>4.30</td>
<td>-3.86</td>
</tr>
<tr>
<td>Mean wind speed (m s\textsuperscript{-1})</td>
<td>1.98</td>
<td>3.88</td>
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5. Conclusions

This was the first study to use the eddy covariance technique in a New Zealand city and the first in an open low-rise subtropical residential area where evergreen vegetation is dominant (vegetation cover fraction: 47%). It is also the first time EC CO$_2$ flux data were linked to $^{14}$CO$_2$ measurements to quantify biogenic and anthropogenic CO$_2$ sources contributing to new knowledge about the potential of combining EC measurements with isotopic analysis for a better understanding about the underlying processes that drive urban CO$_2$ fluxes over the course of a year.

The results of this study have shown that a residential site with a year-round active biosphere has a different CO$_2$ flux pattern to that typically observed in cities in colder climates. CO$_2$ drawdown was clearly visible at midday across all seasons pointing to the importance of the photosynthetic CO$_2$ sink associated with evergreen vegetation with no dormant period. However, the mean CO$_2$ flux remained slightly positive across all
observations, suggesting that CO₂ drawdown was not sufficient to offset anthropogenic and biogenic CO₂ emissions. In line with other urban studies, CO₂ fluxes reached a peak during the morning rush hour when traffic flow was highest. Median nighttime CO₂ fluxes were consistently more positive (larger CO₂ source) than daytime CO₂ fluxes reaching a minimum during winter. ¹⁴CO₂ analysis showed that fossil fuel CO₂ drives the morning peak in both CO₂ flux and mixing ratio when wind was aligned with the residential land cover. However, when wind was from the more vegetated area south-east of the study site, biogenic respiration dominated the morning CO₂ enhancement. Fossil fuel CO₂ was not apparent at night or during mid-afternoon, even in winter. This suggests that anthropogenic CO₂ emissions likely played a minor role at the residential study site. Thus, there were few anthropogenic CO₂ emissions to offset at midday, partly explaining the consistent negative CO₂ fluxes.

The results of this study highlight the importance of vegetation in residential areas to mitigate local CO₂ emissions, particularly in cities with a climate that allows evergreen vegetation to maintain high photosynthetic rates over winter. As urban areas grow, urban planners need to consider the importance of urban greenspace to counteract urban CO₂ emissions. While this study suggests favouring evergreen vegetation, further research is required to quantify the photosynthetic CO₂ uptake potential of individual tree species. This could aid species selection for urban tree planting programmes aimed at climate change mitigation. However, results from the radiocarbon isotope analysis indicate that urban vegetation is also a major natural source of CO₂ due to biogenic respiration. Thus, urban greenspace management may also need to consider practices that minimise biogenic CO₂ emissions.

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Highlights

- First eddy covariance measurements from a subtropical low density urban area
- Midday CO\textsubscript{2} uptake was observed across all seasons
- Radiocarbon isotope measurements show that fossil fuel CO\textsubscript{2} drives morning peak
- Biogenic CO\textsubscript{2} dominant source and sink at night and midday, respectively
- Evergreen vegetation is important for mitigating local CO\textsubscript{2} emissions