ELSEVIER

Chemosphere 59 (2005) 1557–1573

CHEMOSPHERE

www.elsevier.com/locate/chemosphere

Characteristics of roadside air pollution in Korean metropolitan city (Daegu) over last 5 to 6 years: Temporal variations, standard exceedances, and dependence on meteorological conditions

Wan-Kuen Jo [a,*], Jin-Ho Park [b]

[a] *Department of Environmental Engineering, Kyungpook National University, 1370 Sankeok-dong, Bukgu, Daegu 702-701, Korea*
[b] *Environment and Energy Research Center, Research Institute of Industrial Science and Technology, Hyoja-Dong, Nam-Ku, Pohang, Kyungpook 790-330, Korea*

Received 24 June 2004; received in revised form 29 November 2004; accepted 13 December 2004

Abstract

The present study performed a roadside data analysis to provide baseline data for exploring associations between environmental exposure to four gaseous pollutants and health effects on residents living near roadways. The yearly roadside concentrations of CO and SO_2 showed a well-defined decreasing trend, whereas those of NO_2 and O_3 exhibited the reverse trend. In most cases, the diurnal trends of the roadside concentrations were well-defined for all seasons, plus the daytime concentrations were higher than the nighttime concentrations. In contrast to the other target pollutants, the daytime O_3 concentrations observed at the roadside sites were lower than those observed at the residential site, likely due to high-levels of fresh NO from traffic emissions that rapidly react with O_3, thereby reducing the O_3 roadside level. The Sunday roadside concentrations of CO, NO_2, and SO_2 were similar to or somewhat lower than the weekday concentrations. Conversely, for O_3, the Sunday roadside concentrations were similar to or somewhat higher than the weekday concentrations. The higher O_3 concentrations on Sunday may be due to the reduced titration from a decrease in NO_x emissions under VOC-limited conditions (low VOC/NO_x conditions). The monthly averages of O_3 concentrations exhibited the reverse seasonal variation to the other target compounds, with peak O_3 concentrations between April and June, and the second peak between August and October. It is also suggested that for O_3, the 8-h standard is more stringent than the 1-h standard, while for NO_2 and SO_2, the 1-h standard is more stringent than the 24-h standard. The multiple regression equations obtained from the relationship between the concentrations and five meteorological parameters indicated that the number and type of meteorological variables in the equations varied according to the pollutant, monitoring station, or season.

Keywords: NO_2; O_3; CO; SO_2; Multiple regression

* Corresponding author. Tel.: +82 53 950 6584; fax: +82 53 950 6579.
E-mail address: wkjo@knu.ac.kr (W.-K. Jo).

0045-6535/$ - see front matter © 2005 Elsevier Ltd. All rights reserved.
doi:10.1016/j.chemosphere.2004.12.021

1. Introduction

The inference of environmental exposure based on proximity to pollution sources has been used in many epidemiological studies to explore associations between environmental exposure and health effects (Huang and Batterman, 2000). Since motor vehicle emissions are a major source of urban air pollution (Fujita et al., 1992; Derwent et al., 1995; Kodama et al., 2002), it has been suggested that urban residents living near a roadway experience higher exposure to motor vehicle emissions compared to those living further away, thereby experiencing a higher health risk. In fact, certain recent epidemiological studies have confirmed this inference by reporting that residents living near roadways experience more chronic respiratory symptoms, lung function decrements, and more hospital admissions for asthma (Wjst et al., 1993; Edwards et al., 1994; van Wijnen and van der Zee, 1998).

Many residences in Korea front onto roadways with moderate to heavy traffic volumes. As such, this situation has forced Korean environmental policy makers to create a roadside air quality management program to protect nearby residents from potential heath risks. The development of air quality policies usually involves air-monitoring and an analysis of the monitoring data. Therefore, roadside air has been monitored in many Korean cities since 1995. However, a systematic analysis of the monitored roadside air quality data has not yet been reported, although several systematic analyses of general ambient air quality data obtained from fixed monitoring stations in residential, commercial, or industrial areas have been reported (Lee et al., 1999; Jo et al., 2000; Kim et al., 2003). Accordingly, the present study conducted a roadside data analysis to provide baseline data for exploring associations between environmental exposure to major gaseous pollutants and health effects on residents living near roadways, and establishing a roadside air quality management program for the Korean metropolitan city of Daegu. The current data analysis could also be applied to other urban roadside areas in other countries that have similar urban characteristics to Daegu. Hence, the characteristics of roadside CO, NO_2, O_3, and SO_2 are considered over a time period of 5–6 years, along with an analysis of yearly trends, diurnal, weekly, and seasonal variations, an analysis of the frequency of days with concentrations exceeding the Korean ambient air quality standards (KAAQS), and an examination of the relationship with meteorological conditions. The KAAQS are currently set at 1-h/25-ppm and 8-h/9-ppm for CO, 1-h/150-ppb and 24-h/80-ppb for NO_2, 1-h/100-ppb and 8-h/60-ppb for O_3, and 1-h/150-ppb and 24-h/50-ppb for SO_2.

Located in the southern area of Korea and the third largest Korean city, Daegu has a population of ≈2.5 million and population density of 2812/km^2. The Daegu Regional Environmental Management Office (DREMO) estimates that more than one million people in Daegu live within 200 m of major roadways (Jo and Moon, 1999). The city is a low, flat area surrounded by mountains and hills, plus it is located about 80 km away from the nearest sea. These features can serve as barriers that impede the transport of air pollutants to other areas.

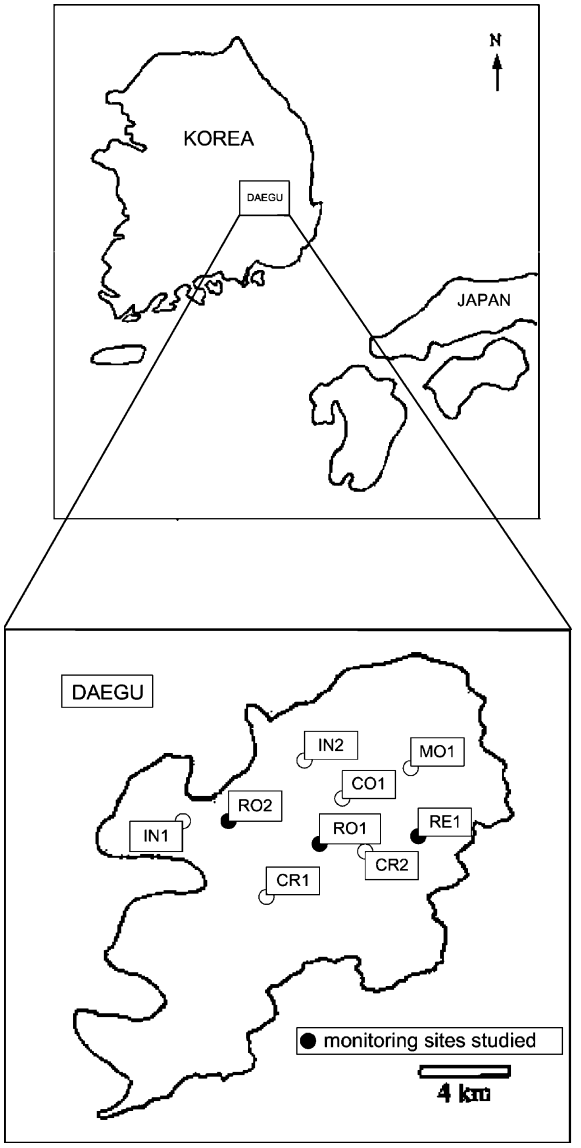

Fig. 1. Monitoring site map: IN, industrial air-monitoring site; CO, commercial air-monitoring site; CR, commercial/residential air-monitoring site; RE, residential air-monitoring site; RO, roadside air-monitoring site; and MO, meteorological observation site. IN1, Jungridong; IN2, Nowondong; CO1, Suchangdong; CR1, Daemyungdong; CR2, Samdeokdong; RE1, Manchondong; RO1, Namsandong; RO2, Pyungridong; MO1, Sinamdong.

Moreover, many roadways are surrounded by high-story buildings. Based on the statistical yearbook for Daegu published in August 2003, the number of vehicles registered in the Daegu metropolitan area was as follows: 495 562 passenger cars, 20 098 taxicabs, 62 284 buses (3260 public buses), 152 297 trucks, 108 585 motorcycles, and 978 special cars. Traffic movement in the city is typically slow during the morning and evening rush-hour periods. Most of the passenger cars and motorcycles consume gasoline, while the buses and taxis generally use diesel-fuel and liquid petroleum gas (LPG), respectively.

2. Methodology

The DREMO and Daegu municipal government operate eight monitoring stations for KAAQS: two at industrial sites, one at a commercial site, two at commercial/residential sites, one at a residential site, and two at roadside sites (Fig. 1). The concentrations of target compounds were measured using conventional techniques: CO, non-dispersive infrared method; NO_2, chemiluminescence method; O_3, ultraviolet (UV) photometric method and SO_2, pulse UV fluorescence method. Air data from the two roadside monitoring stations (RO1 and RO2) were analyzed in the present study, along with air data from the residential monitoring station (RE1), located 700 m away from a major roadway, that was considered as the control station. RO1 is situated 1 m from the curbside of the Daero, which is a ten-lane main roadway (five lanes in each direction) and one of the major arterial roadways in the city. Meanwhile, RO2 is located 1 m from the curbside of the Taepyungro, which is a six-lane roadway. Both roads connect the west and east sections of Daegu and cross the downtown with heavy traffic. The distance between the two roadside sites is ≈3 km. Traffic congestion is a common occurrence alongside the roadside monitoring sites. RE1 is situated ≈8 km to the east of RO2.

A mobile container was set up for each roadside air-monitoring station with roof-top sampling inlets at a height of ≈2.5 m above the level of the road. Whereas, the residential air-monitoring was performed on the roof

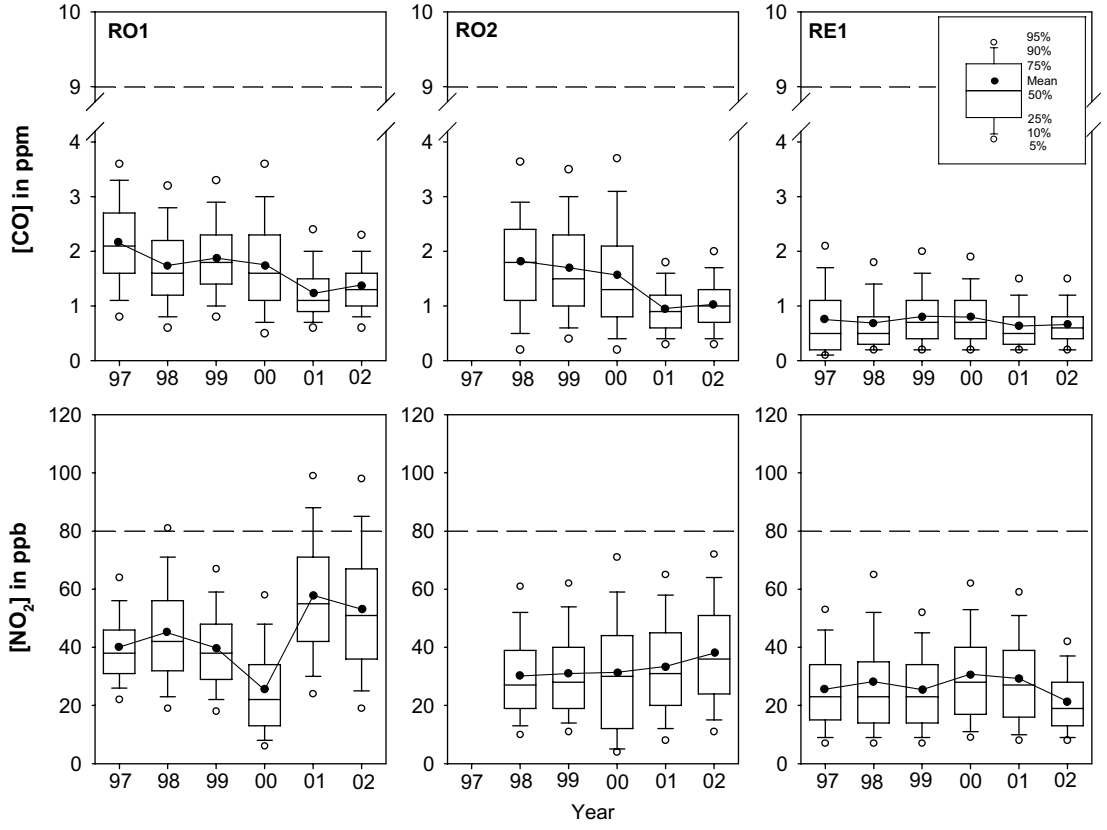

Fig. 2. Yearly trend of 1-h average CO and NO_2 concentrations in roadside and residential areas during 5- or 6-year study period. The box plot represents the 95‰, 90‰, 75‰, 50‰, 25‰, 10‰, and 5‰. The mean values (●) are joined by lines. The broken line indicates the current 8-h KAAQS for CO (9 ppm) and current yearly KAAQS for NO_2 (50 ppb).

of a three-story elementary school at a height of about 9 m above the ground. Twelve readings per hour (one per 5 min) are typically used to calculate a 1-h average concentration. Once the data have been quality assured, the information is then transferred to the Air Quality Management Bureau (AQMB) of the Korean Ministry of Environment. The data presented in this study, comprising 1-h averaged concentrations, were extracted from the AQMB database. The current analysis makes use of the 1-h averaged concentrations of CO, NO_2, O_3, and SO_2 measured at the selected monitoring stations, and the study begins with data from 1997 or 1998, marking the initiation of quality assured data. Data was only included in the study when 20 or more hourly values were available during each 24-h period at all of the selected stations. Meanwhile, the meteorological data employed were obtained from the Daegu meteorological office (MO1), situated 6 km east of RO2 (Fig. 1). Hourly observations of solar radiation, wind speed, temperature, relative humidity, and precipitation were obtained for the study period. The statistical analyses included an *F*-test and stepwise multiple regression procedures using the SAS program (Version 8) on a personal computer.

3. Results and discussion

3.1. Yearly trends

General trends were evaluated using the 1-h average concentrations of the four compounds during the 5- or 6-year study period. Box plots of the 1-h average concentrations of the CO, NO_2, O_3, and SO_2 concentrations monitored at the two roadside sites and one residential site were generated as shown in Figs. 2 and 3. In addition, the percentage changes in the mean and 50th and 95th percentiles were computed by fitting a linear relationship to the annual means and dividing the slope of the lines by the overall average (Table 1) (Fuentes and Dann, 1994). The reported concentration changes were then referenced to the overall annual mean values. As presented in Fig. 2, the mean and percentiles for the CO concentrations all showed a well-defined decreasing

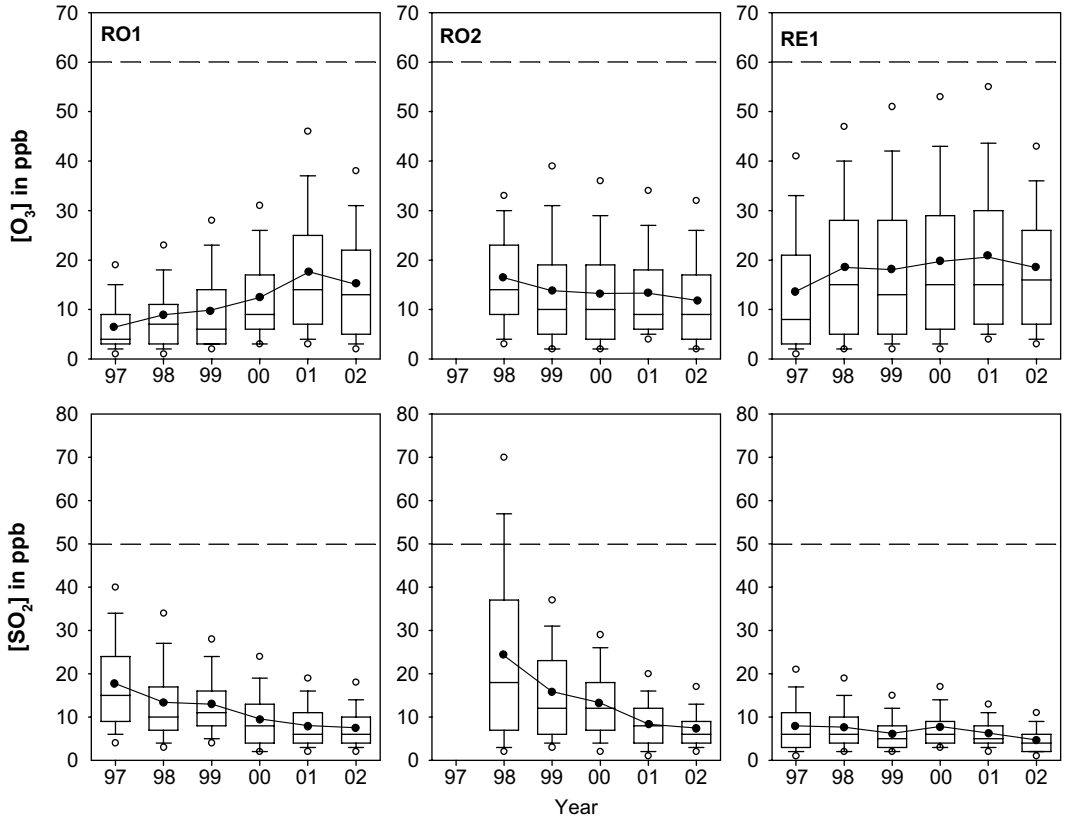

Fig. 3. Yearly trend of 1-h average O_3 and SO_2 concentrations in roadside and residential areas during 5- or 6-year study period. The box plot represents the 95‰, 90‰, 75‰, 50‰, 25‰, 10‰, and 5‰. The mean values (●) are joined by lines. The broken line indicates the current 8-h KAAQS for O_3 (60 ppb) and current yearly KAAQS for SO_2 (20 ppb).

Table 1
Changes in annual mean and percentiles of 1-h concentrations of CO, NO_2, O_3, and SO_2 at two roadside monitoring sites and one residential monitoring site during 5- or 6-year study period[a]

Monitoring site	Compound	Mean		50th		95th	
		Δ	*F*-test	Δ	*F*-test	Δ	*F*-test
RO1	CO	−9.4	**	−10.1	**	−8.2	**
	NO_2	+5.8	ns	+6.1	ns	+7.9	ns
	O_3	+17.4	**	+22.3	**	+16.0	**
	SO_2	−17.5	**	−18.3	**	−16.7	**
RO2	CO	−17.1	**	−16.9	**	−16.8	**
	NO_2	+5.5	**	+6.9	**	+3.8	ns
	O_3	−7.1	**	−10.6	*	−2.0	ns
	SO_2	−30.0	**	−25.0	**	−35.5	**
RE1	CO	−2.7	ns	+1.7	ns	−6.1	**
	NO_2	+1.4	ns	−0.4	ns	−3.2	ns
	O_3	+5.2	ns	+9.6	**	+2.1	ns
	SO_2	−8.0	*	−6.4	ns	−11.8	**

[a] Δ indicates the changes in annual mean and percentiles; $\Delta = (\text{slope}/\mu)100\%$, where μ = overall annual mean; ns, not statistically significant; * Significant with $p < 0.1$; ** significant with $p < 0.05$; and calculations were made using the USEPA technique (USEPA, 1995).

trend at the two roadside stations (RO1 and RO2) over the period from 1997 or 1998 to 2002, plus the changes in the mean and 50th and 95th percentiles were all statistically significant with $p < 0.05$ (Table 1). The mean

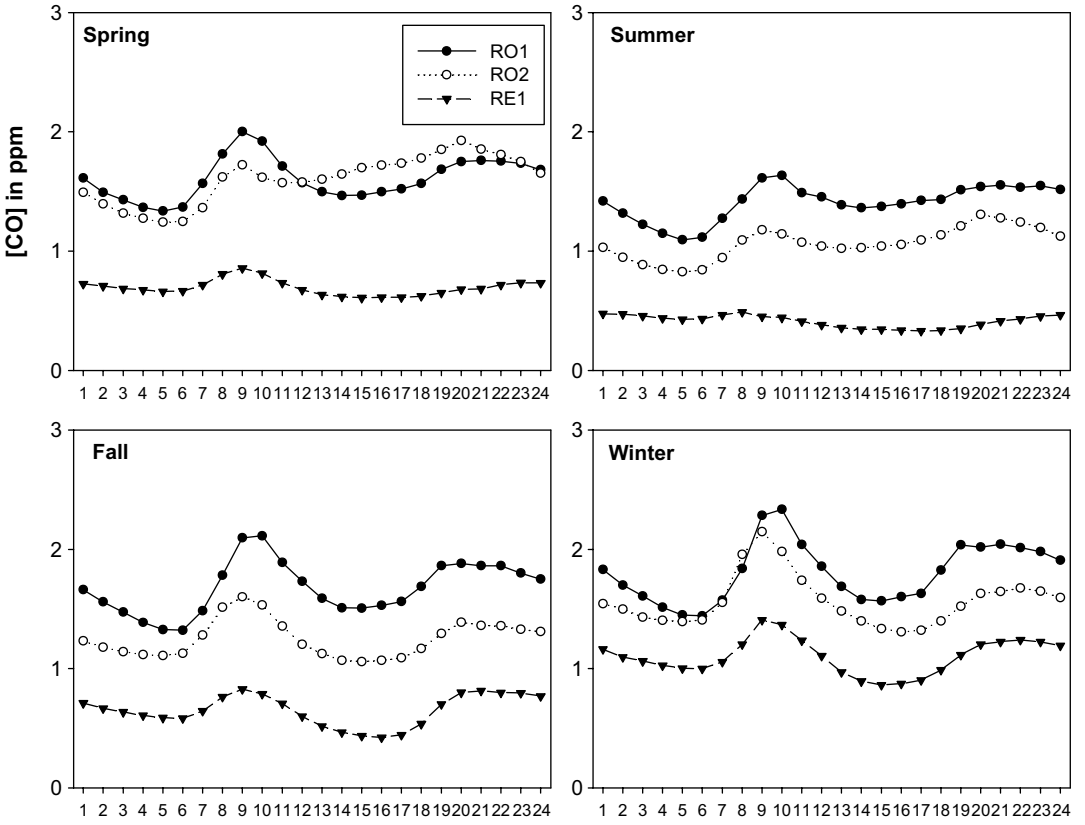

Fig. 4. Diurnal CO averages in roadside and residential areas for four seasons.

concentrations at RO1 changed from 2.2 (1997) to 1.7 ppm (2002) over the 6-year study period, while those at RO2 changed from 1.9 (1998, starting year) to 0.8 ppm (2002) over the 5-year study period. However, at the residential site (RE1), the 95th percentile (−6.1%) showed a lower decreasing rate compared to the 95th percentiles for RO1 (−8.2%) and RO2 (−16.8%). According to the Daegu statistical yearbooks, the total number of motor vehicles registered in the Daegu metropolitan area gradually increased from 584914 (1997) to 785475 (2002). This increase in the vehicle number potentially should have elevated the CO levels at the roadways, since motor vehicle emissions (91%) are the largest source of CO in Korea (Lee et al., 1999), yet the present results show the reverse. Moreover, none of the percentile concentrations at either the roadside or residential sites exceeded the 8-h/9-ppm standard during the 6-year study period. In fact, the highest 95th percentiles during the study years were 3.7 (1997), 3.5 (1998), and 2.3 ppm (1997) at RO1, RO2, and RE1, respectively. Consequently, it would appear that CO pollution was not recently significant at the monitored roadside and residential sites.

In contrast to CO, the mean and both percentiles of the NO_2 concentrations showed an increasing trend at the two roadside stations (RO1 and RO2) (Fig. 2), although only the mean and 50th percentile at RO2 were statistically significant (Table 1). The mean concentrations at RO1 changed from 24 (2000) to 56 ppb (2001), while those at RO2 changed from 19 (1998) to 37 ppb (2002). Moreover, the 95th percentile concentrations at the two roadside sites were close to or exceeded the Korean year/50-ppb standard. In particular, the 95th percentiles at RO1 for the two most recent years (2001 and 2002) were 97 and 95 ppb, respectively. As such, attention should be given to the NO_2 exposure of residents living near roadways. Meanwhile, no notable trend was observed at RE1.

Similar to NO_2, the mean and both percentiles of the O_3 concentrations showed a well-defined increasing trend at RO1 over the 6-year study period (Fig. 3). The changes in the mean and percentiles were all statistically significant with $p < 0.05$ (Table 1). The mean concentrations at RO1 changed from 7 (1997) to 16 ppb (2001). The residential site (RE1) also exhibited an increasing trend. Although none of the 95th percentiles during the 5- or 6-year study period exceeded the Korean 8-h/60-ppb standard, the increasing roadside trend still suggests that attention should be given to

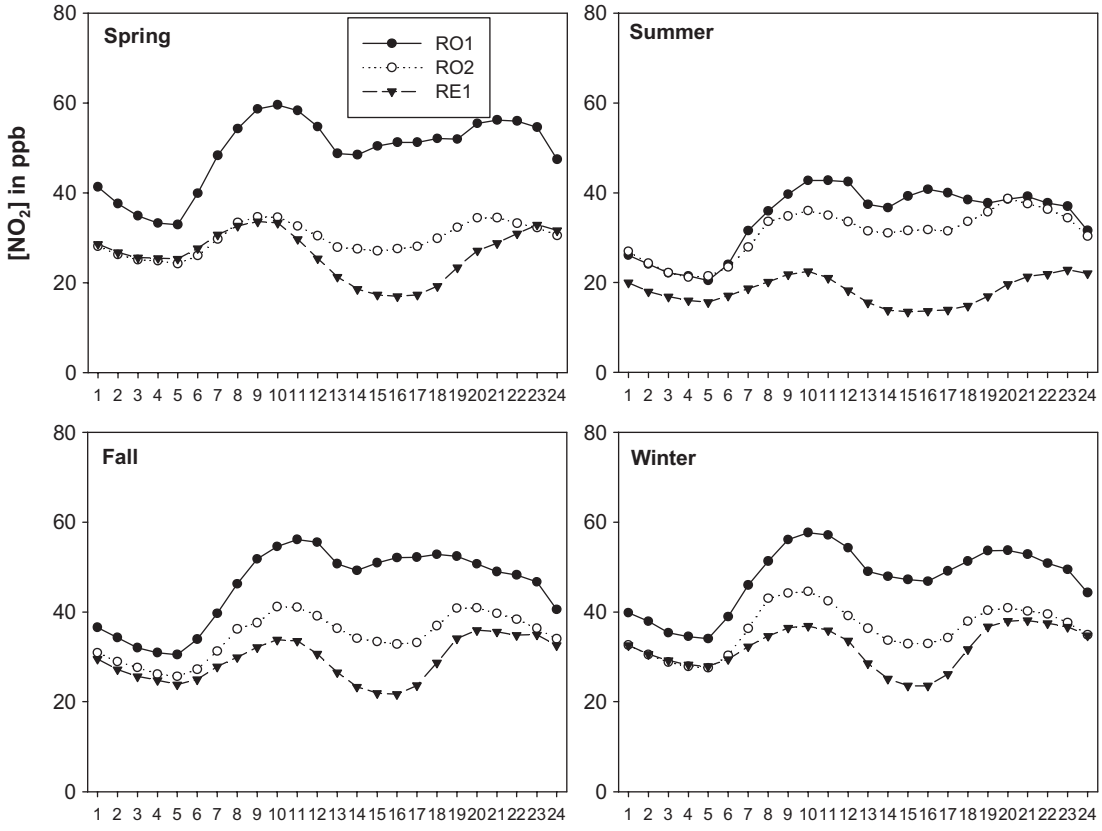

Fig. 5. Diurnal NO_2 averages in roadside and residential areas for four seasons.

the O_3 exposure of residents living near roadways. As expected, the trends could be ascribed to the combined effects of emission inventories of O_3 precursors and meteorological conditions (NRC, 1991; Davidson, 1993; Ghim and Chang, 2000; Ghim et al., 2001). Conversely, for RO2 the mean and both percentiles of the O_3 concentrations showed a decreasing trend over the 6-year study period (Fig. 3), although the 95th percentile was not statistically significant (Table 1). Unfortunately, this trend is unexplainable, as the necessary information was unobtainable.

For SO_2, the mean and both percentiles of the concentrations showed a well-defined decreasing trend at the two roadside sites over the 6-year study period (Fig. 3). The changes in the mean and percentiles were all statistically significant with $p < 0.05$ (Table 1). The mean concentrations at RO1 changed from 18 (1997) to 6 ppb (2002), while those at RO2 changed from 23 (1998) to 8 ppb (2002). Similarly, RE1 exhibited a decreasing trend for the SO_2 concentrations. Moreover, the 95th percentile in the most recent year seldom exceeded the Korean year/20-ppb standard, suggesting that SO_2 pollution was not recently significant at the monitored roadside and residential sites.

3.2. Diurnal variations

Diurnal variations in urban air pollutant concentrations attract great concern, since the changes in activity and emissions that occur each day and night provide a natural experiment. Accordingly, the current study analyzed the diurnal variations of the four target compounds during the four different seasons over the 5- or 6-year study period. As shown in Fig. 4, the diurnal CO trends for both the roadside and residential stations were very similar for all four seasons. The diurnal averages exhibited a gradual increase in the pollution levels, commencing around 07:00 h (the start of morning traffic) and culminating at approximately 10:00 h. The concentration then proceeded to a shallow trough before increasing once again to give a second peak concentration around 19:00–20:00 h (evening traffic period). Thereafter, the concentration of CO remained almost steady. This diurnal CO trend is similar to the diurnal trends of other urban areas abroad (Fujita et al., 2003; Mönkkönen et al., 2004). Consequently, the daytime CO concentrations were higher than the nighttime concentrations. This result is also consistent with a previous urban study conducted in India (Mönkkönen et al.,

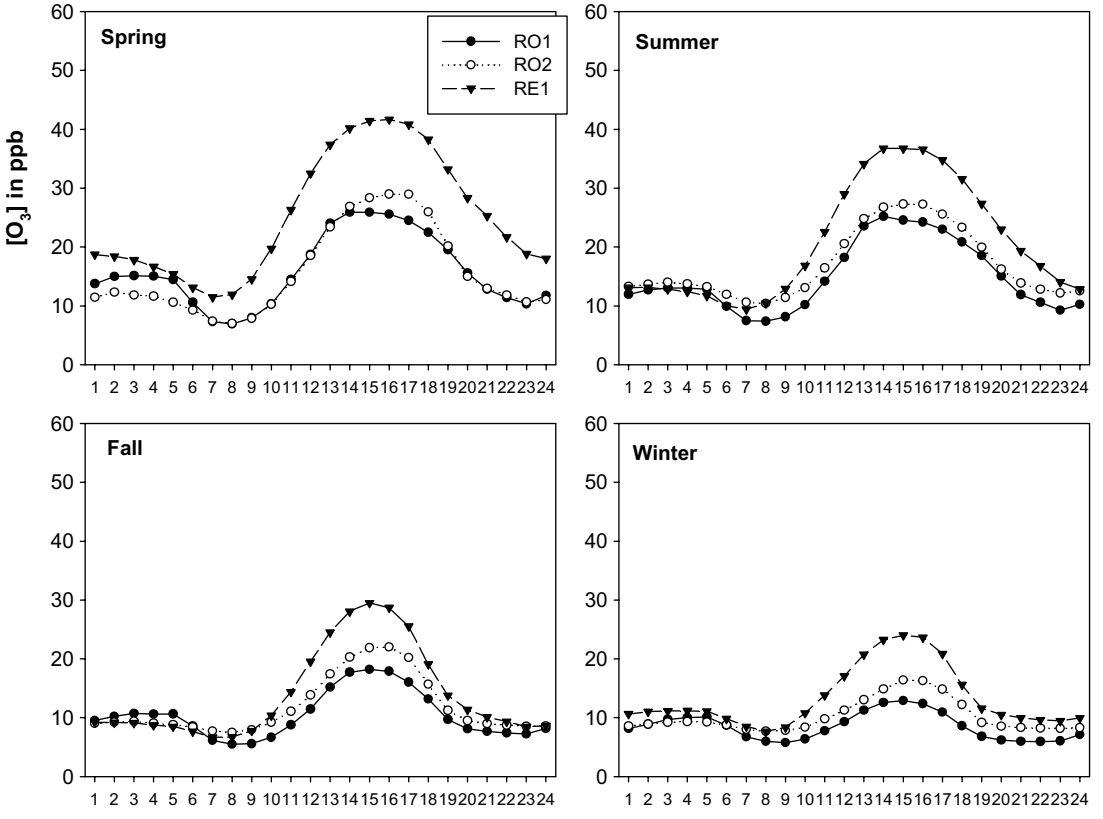

Fig. 6. Diurnal O_3 averages in roadside and residential areas for four seasons.

2004), which reported that the evening concentrations were usually higher presumably due to the extended domestic use of fossil fuels, transportation, and biofuels in the evenings. Meanwhile, the maximum roadside hourly mean concentration was observed at RO1 in the winter (2.4 ppm, 10:00 h), and the minimum observed at RO2 in the summer (0.8 ppm, 05:00 h). The roadside CO concentrations were higher than the residential concentrations throughout the diurnal hours.

Similar to CO, the diurnal NO_2 trends at RO2 and RE1 were well-defined for all seasons, whereas no clear trends were found at RO1 (Fig. 5). The diurnal NO_2 averages at RO2 and RE1 exhibited a gradual increase in the pollution levels, commencing around 06:00–07:00 h and culminating at approximately 09:00–11:00 h. The concentration then proceeded to a shallow trough before increasing once more to give a second peak concentration round 19:00–21:00 h. Thereafter, the concentration decreased steadily until around 05:00 h. This diurnal NO_2 trend is similar to the diurnal trend found at Azusa in California's South Coast Air Basin (Fujita et al., 2003). Meanwhile, the maximum roadside hourly mean concentration was observed at RO1 in the spring (60 ppb, 10:00 h) and the minimum observed at RO1 and RO2 in the summer (19 ppb, 05:00 h). The maximum concentration in the spring was similar to that in the fall and winter, and the minimum concentration in the summer similar to that in the other seasons. A relative maximum in the NO_2 levels in RO1 shown between 14:00 and 18:00 h in all seasons except winter. This increase is registered during the hours of maximal boundary layer depth, and thus the source must be significant as it counters the dilution of pollutants in a larger volume during these hours. Generally, the roadside NO_2 concentrations were higher than the residential concentrations throughout the diurnal hours. The only exception was that the NO_2 concentrations observed late at night and early in the morning (from 23:00 to 06:00 or 10:00 h) during three seasons (spring, fall and winter) were similar for the residential (RE1) and roadside (RO2) stations.

The diurnal patterns for O_3 were different from those for CO, and similar at all the monitoring stations for all seasons (Fig. 6). The first peak O_3 concentrations were observed about 5 h later than those for NO_2 (15:00 h). The diurnal averages exhibited a gradual increase in the pollution levels, commencing around 09:00 h and culminating at approximately 15:00 h. Thereafter, the

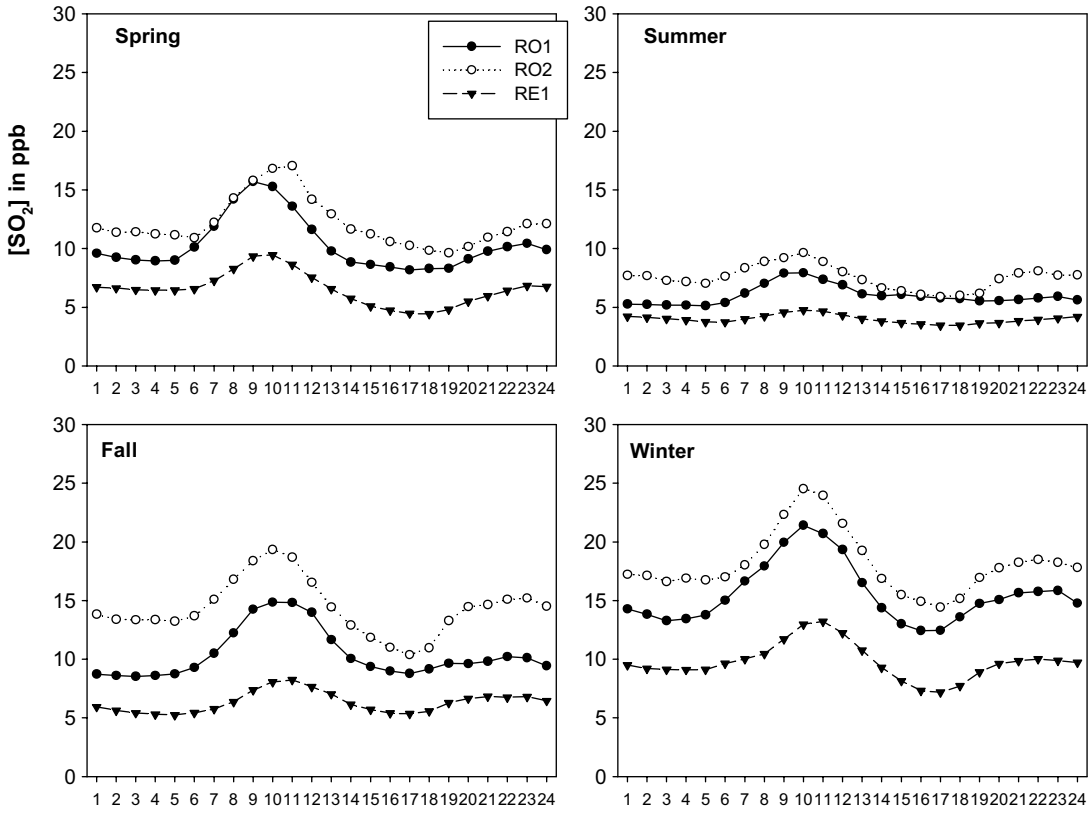

Fig. 7. Diurnal SO_2 averages in roadside and residential areas for four seasons.

concentration decreased steadily until around 20:00 h, then remained almost steady. This diurnal trend is similar to the diurnal trends found in several other US cities (Fujita et al., 2003; Pun and Seigneur, 2003). Meanwhile, the maximum roadside hourly mean concentration was observed at RO2 in the spring (29 ppb, 16:00 h) and the minimum observed at RO1 in the fall and summer (4 ppb, 08:00 h for fall and 09:00 h for winter). The maximum concentration in the spring was similar to that in the summer, yet higher than that in the fall and winter. Besides the afternoon hours, including the peak hours, the O_3 concentrations were very similar at the two roadside sites (RO1 and RO2). Contrary to the other target pollutants, the daytime O_3 concentrations observed at the roadside sites were lower than those at the residential site. The reason for this is described in a later section.

Similar to CO, the diurnal SO_2 trends for both the roadside and residential stations were well-defined for all seasons (Fig. 7). The diurnal averages exhibited a gradual increase in the pollution levels, commencing around 07:00 h and culminating at approximately 10:00 h. The concentration then proceeded to a shallow trough before increasing once more to give a second peak concentration around 19:00–20:00 h. Thereafter, the concentration remained almost steady. Besides the morning and evening traffic hours, there was no significant difference in the SO_2 concentrations between the daytime and nighttime hours. This result is not consistent with a previous study conducted in India (Mönkkönen et al., 2004). Fig. 7 shows peak SO_2 levels coinciding with the rush hours, the same as occurs with NO_2. This situation is supported by the fact that in Korea, motor vehicle emissions are still an important source for SO_2, the third largest source after heating and industry (Lee et al., 1999). Meanwhile, the maximum roadside hourly mean concentration was observed at RO2 in the winter (25 ppb, 10:00 h) and the minimum observed at RO2 in the summer (5 ppb, 05:00 h). The roadside SO_2 concentrations were higher than the residential concentrations throughout the diurnal hours.

3.3. Day-of-the-week variations

Similar to diurnal variations, since the changes in activity and emissions that occur each week provide a natural experiment (Fujita et al., 2003; Heuss et al., 2003; Judith, 2003), the present study analyzed the

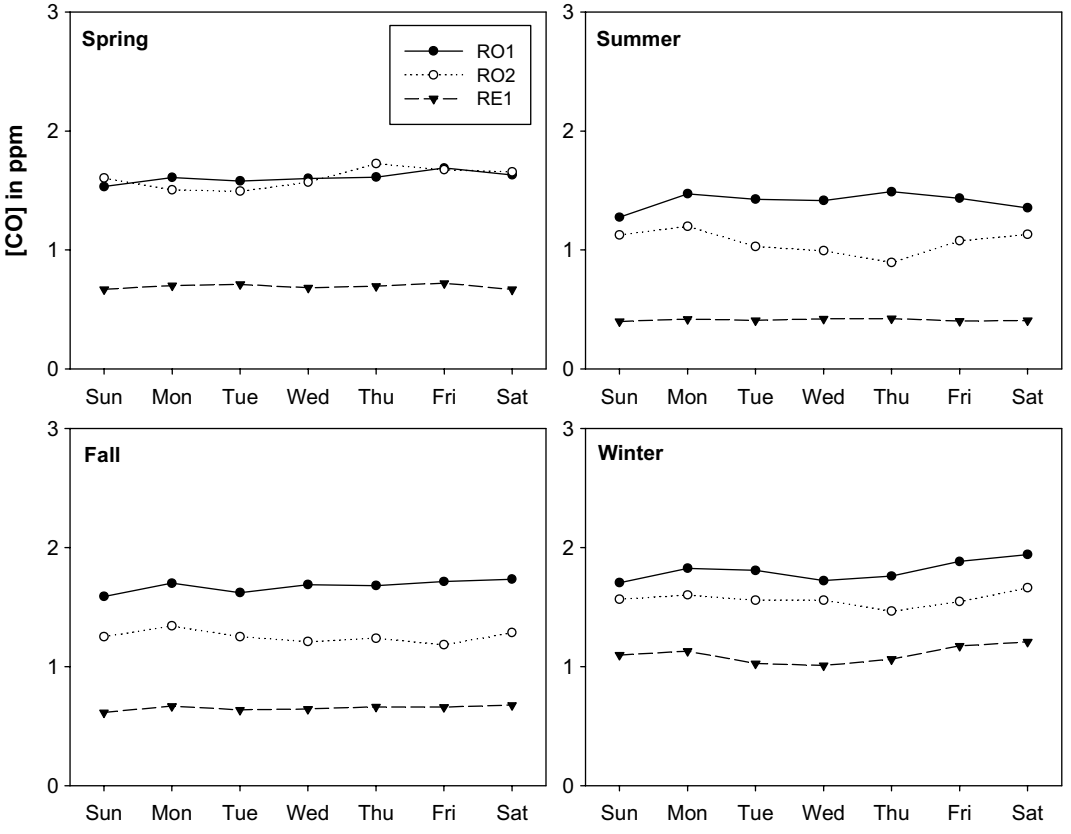

Fig. 8. Day-of-the-week variation of 1-h average concentrations of CO in roadside and residential areas for four seasons.

day-of-the-week variations using the 1-h average concentrations of the four target compounds for the four different seasons (Figs. 8–11). The Sunday roadside concentrations of CO, NO_2, and SO_2 were similar to or somewhat lower than the weekday concentrations. Conversely, for O_3 the Sunday roadside concentrations were similar to or somewhat higher than the weekday concentrations. The roadside weekly patterns were also similar to the residential weekly patterns. These higher patterns of O_3 and lower patterns of other compounds on Sunday are consistent with the day-of-the-week variations found in many US cities (Blanchard and Tanenbaum, 2003; Fujita et al., 2003; Heuss et al., 2003). Earlier US studies reported that the weekday/weekend differences are largely caused by differences in activity/emission patterns. During weekends, the emissions of anthropogenic air pollutants are believed to decrease from weekday values, because the major sources of air pollutants, such as motor vehicle traffic and industrial activities, are less active on weekends (Chinkin et al., 2003; Pun and Seigneur, 2003). As such, the previous finding suggests that higher O_3 levels occur, even though the emissions of O_3 precursors, nitrogen oxides (NO_x), and volatile organic compounds (VOC) are lower on weekends than on weekdays (Blanchard and Tanenbaum, 2003; Fujita et al., 2003; Heuss et al., 2003), because of a reduced titration due to a decrease in NO_x emissions under VOC-limited conditions (low VOC/NO_x conditions). The corollary of this interpretation is that under NO_x-limited conditions (high VOC/NO_x conditions), reducing NO_x reduces O_3. Accordingly, the higher O_3 levels on Sunday observed in the present study may be caused by the former phenomenon. Therefore, if this O_3 chemistry interpretation is the primary explanation for the current weekend O_3 effect, policy-makers can employ this result to establish a control strategy for the reduction of O_3 precursors in the study area. However, since other factors may also play a role in the weekend O_3 behavior, a further study is recommended to confirm the present interpretation.

3.4. Seasonal variations

Fig. 12 shows the monthly averages of the 1-h mean concentrations computed for the 5- or 6-year study period. The monthly averages of roadside air pollution may be employed to establish a fuel policy for motor vehicles in different seasons to improve urban air quality, since

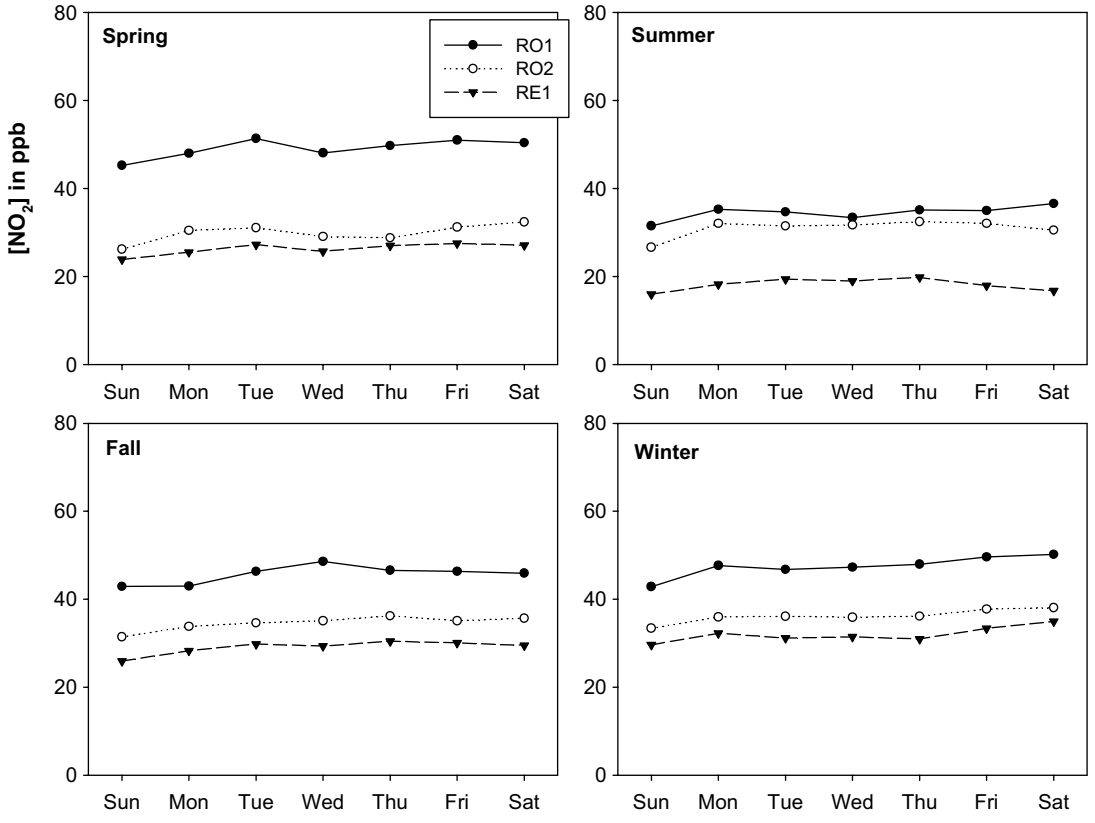

Fig. 9. Day-of-the-week variation of 1-h average concentrations of NO_2 in roadside and residential areas for four seasons.

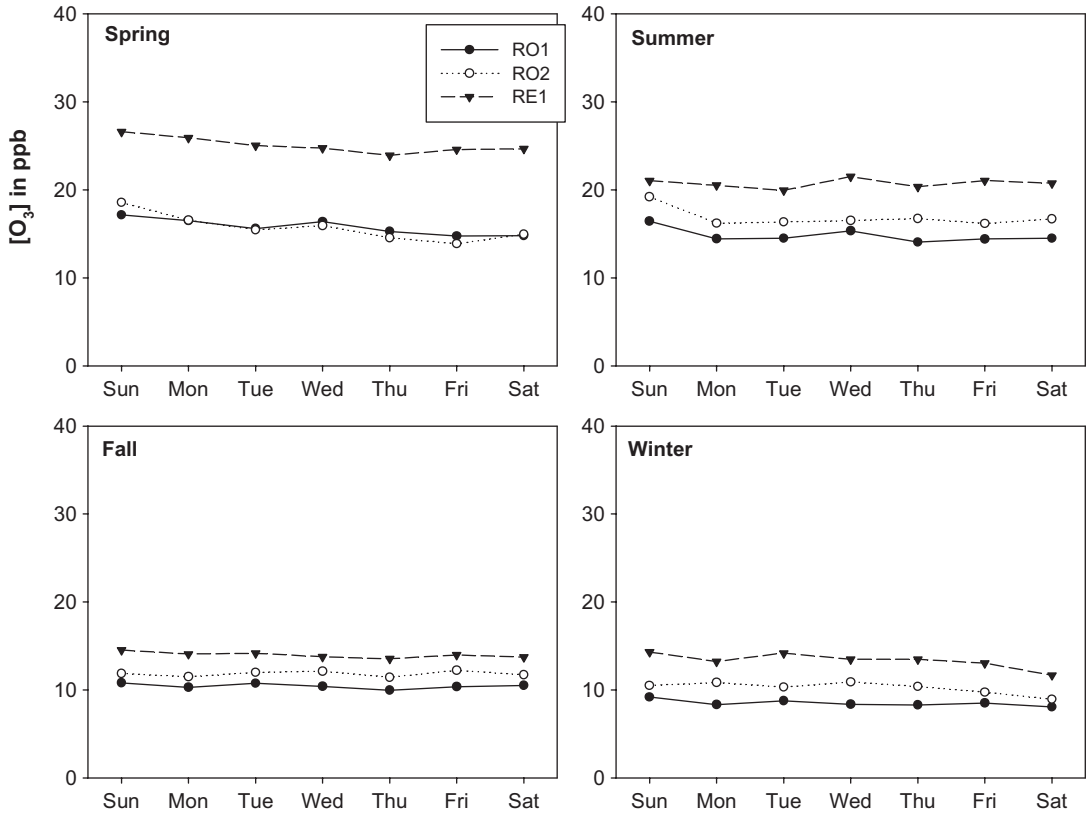

Fig. 10. Day-of-the-week variation of 1-h average concentrations of O_3 in roadside and residential areas for four seasons.

these averages partially reflect the characteristics of motor vehicle emissions for each season. Both CO and SO_2 showed a well-defined seasonal variation in the study city. The seasonal pattern for the concentrations of these two pollutants was similar at the two roadside stations, and at the residential station. The concentrations increased during the winter months, then gradually diminished with the minimum concentrations found between June and September, and finally increased again until the winter months. This pattern is similar to the results of previous studies performed in residential areas in Daegu (Jo et al., 2000) and Seoul, the capital of Korea (Lee et al., 1999). More heating fuels are typically consumed in Korean urban areas during the wintertime, thereby elevating ambient air levels of CO and SO_2 in urban areas, including residential and roadside sites (Lee et al., 1999). An additional cause of the seasonal pattern of CO concentrations found at the roadway sites may have been the lower combustion efficiency of motor vehicle fuels in the winter, thereby causing higher CO concentrations in the roadway air. This is supported by Bruetsch's study (1981), which reported that high-tailpipe emissions from motor vehicles are associated with a cold ambient temperature. Moreover, although the wind speeds varied little between the seasons, other meteorological parameters, such as the temperature and solar radiation, pointed to poorer mixing during the wintertime (Table 2) (Derwent et al., 1995). Meanwhile, for all matched months, the roadside concentrations of the two compounds were higher than the residential concentrations. This is supported by the fact that motor vehicle emissions are the largest source for CO and the third largest source for SO_2 after heating and industry (Lee et al., 1999).

The seasonal variation of the NO_2 concentrations monitored at one roadside station (RO1) and the residential station (RE1) was similar to that of the CO and SO_2 concentrations, although the variation at RO2 did not show any distinct trend. The NO_2 concentrations were lower in the summer months than in the other months. This is likely due to the lower fuel combustion and greater photochemical reaction of NO_2 in the summer months compared to the other months (Jo et al., 2000). Meanwhile, the NO_2 trend at the RO2 site shows a clear summer maximum, although this trend is unexplainable, as the necessary information was unobtainable. Similar to CO and SO_2, for all matched months, the roadside NO_2 concentrations were higher

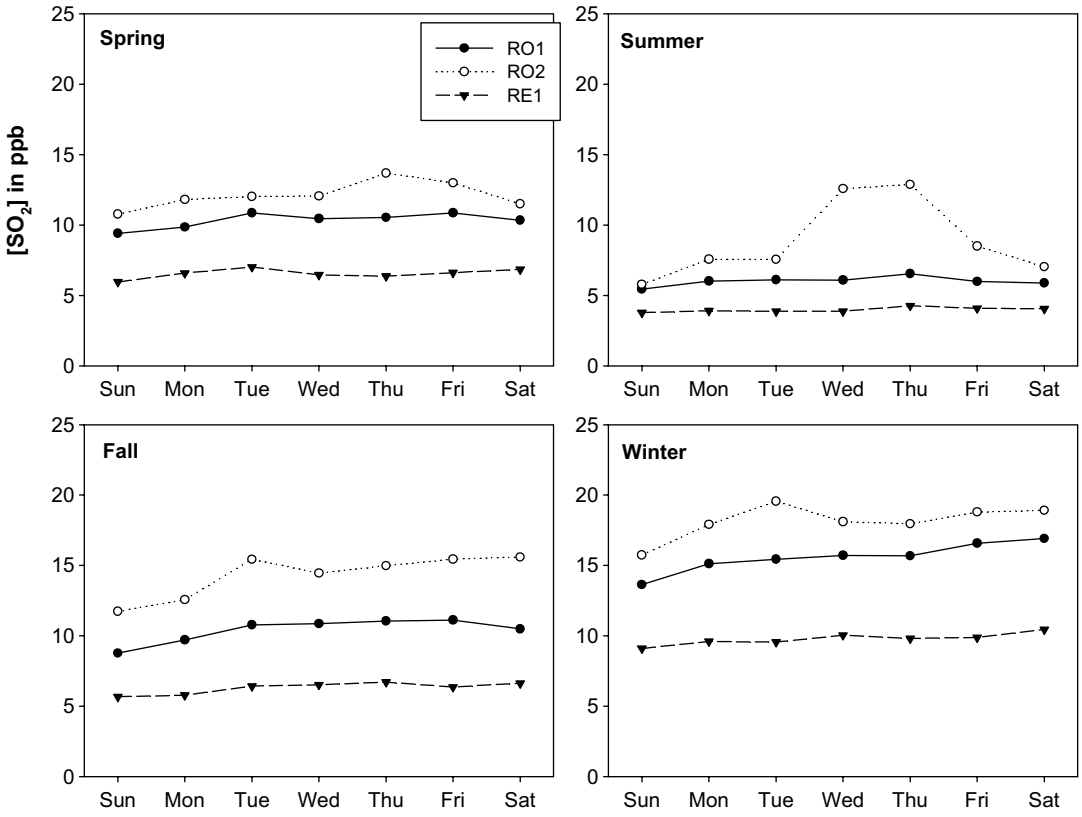

Fig. 11. Day-of-the-week variation of 1-h average concentrations of SO_2 in roadside and residential areas for four seasons.

than the residential concentrations. This is supported by the fact that motor vehicle emissions are the largest source for NO_2 (Lee et al., 1999).

However, the monthly averages of O_3 concentrations exhibited the reverse seasonal variation to the other target compounds. Both the roadside and residential O_3 concentrations slowly increased during the late winter, with the peak attained between April and June and the second peak attained between August and October. The values then gradually diminished, with the minimum concentrations found between December and January for all the monitoring stations. This seasonal pattern reflects the contribution of photochemically generated O_3 from sunshine and anthropogenic and natural precursors. A similar seasonal pattern to the present study was also found over much of the United States and Europe (Logan, 1986; Feister and Balzer, 1991; Alonso et al., 2000). All the current monitoring sites exhibited lower O_3 concentrations in July and August compared to the other summer month, June. This is most likely due to the seasonal rain front that usually occurs in Korea between July and August. The seasonal evolution of O_3 levels was parallel to that of NO_2 in spring–summer, and the inverse in winter, which is the usual pattern at many urban stations (Viana et al., 2003). It has been reported that O_3 levels in urban areas reach high-values on spring and summer days, usually under the influence of stagnant high-pressure systems (Aneja et al., 2000), as characterized by the summer bright sunshine and high-temperatures (Table 2). Contrary to the other target compounds, for all matched months, the roadside O_3 concentrations were lower than the residential concentrations. This result is likely due to high-levels of fresh NO from traffic emissions, which rapidly react with O_3, thereby reducing the O_3 roadside level (Yang and Miller, 2002).

3.5. Standard exceedances

To analyze the frequency of elevated concentrations over the study period, the number of days when the concentrations of the target pollutants exceeded the corresponding 1-h, 8-h, or 24-h standards was totaled. The frequencies were then determined by calculating the total number of days with concentrations exceeding the corresponding standard as a percentage of the total number of sampling days over the specified time period (Table 3). No days exceeding the 1-h CO standard were

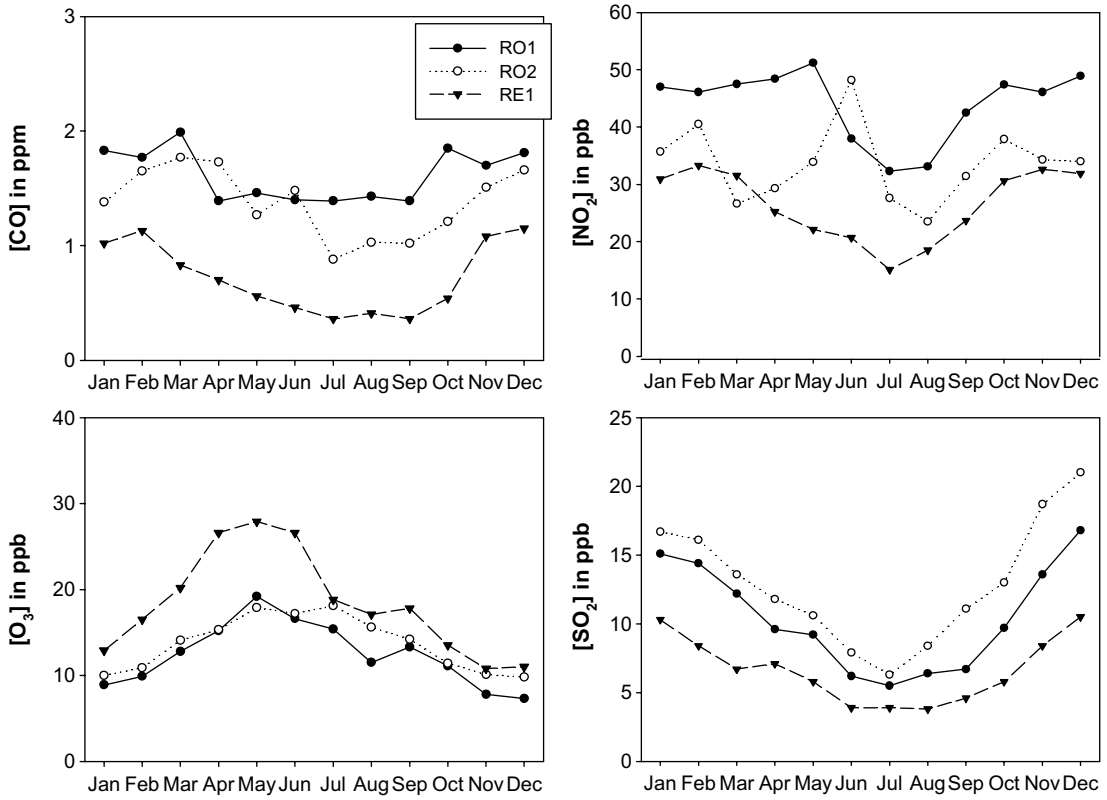

Fig. 12. Monthly variation of 1-h average concentrations of CO, NO_2, O_3, and SO_2 in roadside and residential areas.

Table 2
Seasonal average values for meteorological data

Season	Solar radiation (MJ/m^2)	Wind speed (m/s)	Temperature (°C)	RH (%)	Precipitation (mm)
Spring	15	2.6	17	53	2.0
Summer	25	2.3	15	70	7.2
Fall	16	2.0	11	64	3.1
Winter	3	2.5	9	53	0.7

observed at any of the monitoring sites, while the frequency of days with CO concentrations above the 8-h standard ranged from 0.1% to 0.4%. However, for the other target pollutants the frequency of days with concentrations above the 1-h standard ranged from 0.5% to 4.3%, while the frequency of days above the 8- or 24-h standard ranged from 0.3% to 5.1%. It should be noted that the frequency of days with O_3 concentrations above the 8-h standard at the residential site (RE1) was five times higher than that at the roadside sites (RO1 and RO2).

The frequency of days with O_3 concentrations above the 8-h standard was greater than the frequency of days above the 1-h standard at all sites. Conversely, the frequency of days with NO_2 and SO_2 concentrations above the corresponding 24-h standard was less than the frequency of days above the corresponding 1-h standard at all sites. Thus, it would seem that, for O_3, the 8-h standard is more stringent than the 1-h standard, while for NO_2 and SO_2, the 1-h standard is more stringent than the 24-h standard.

3.6. Relationship with meteorological conditions

To examine the possible relationships between the pollutant concentrations and various meteorological parameters, stepwise multiple regression analyses were performed between the air concentrations as the dependent variable and meteorological parameters as independent variables. The meteorological parameters included the daily average solar radiation, wind speed, temperature, relative humidity and precipitation, all of which are associated with urban air pollution (particularly, O_3) levels (Broomfield et al., 1996; Krupa et al., 2003). Table 4(Panels A and B) show the regression equations relating the air concentrations with the meteorological data, which were chosen according to the coefficient of determination (R^2) and Mallows' *Cp* value (Draper and Smith, 1981). The number of and type of

Table 3
Frequency of days with CO, NO_2, O_3, and SO_2 concentrations exceeding 1-h and 8-h or 24-h standards at two roadside monitoring sites and one residential site during 5- or 6-year study period[a]

Monitoring site	No. of years observed	Compound	Frequency of days (%)		
			1-h standard[b]	8-h standard[c]	24-h standard[d]
RO1	6	CO	0.0	0.4	NA
		NO_2	2.5	NA	0.6
		O_3	0.6	0.9	NA
		SO_2	4.3	NA	1.7
RO2	5	CO	0.0	0.4	NA
		NO_2	0.6	NA	0.4
		O_3	0.5	0.9	NA
		SO_2	3.2	NA	1.3
RE1	6	CO	0.0	0.1	NA
		NO_2	0.5	NA	0.3
		O_3	0.7	5.1	NA
		SO_2	1.5	NA	0.7

[a] NA: not available.
[b] 1-h standard: 25 ppm, CO; 150 ppb, NO_2; 100 ppb, O_3; 150 ppb, SO_2.
[c] 8-h standard: 9 ppm, CO; 60 ppb, O_3.
[d] 24-h standard: 80 ppb, NO_2; 50 ppb, SO_2.

Table 4
Regression analysis of CO (ppm) and NO_2 (ppb) [Panel A] and O_3 (ppb) and SO_2 (ppb) [Panel B] concentrations and meteorological data according to monitoring site and season[a]

Composition	Site	Season	Regression equation	R^2	P
Panel A					
CO	RO1	Spring	$\text{Conc} = 2.6(\pm0.2) - 0.02(\pm0.01)X_1 - 0.2(\pm0.04)X_2 - 0.02(\pm0.01)X_3$	0.10	0.0024
		Summer	$\text{Conc} = 1.8(\pm0.08) - 0.01(\pm0.00)X_1 - 0.08(\pm0.02)X_2$	0.08	0.0012
		Fall	$\text{Conc} = 2.9(\pm0.3) - 0.05(\pm0.01)X_1 - 0.2(\pm0.035)X_2 + 0.02(0.01)X_3 - 0.01(\pm0.00)X_4$	0.17	0.0466
		Winter	$\text{Conc} = 2.4(\pm0.09) - 0.2(\pm0.03)X_2$	0.14	0.0001
	RO2	Spring	$\text{Conc} = 2.7(\pm0.3) - 0.2(\pm0.06)X_2 - 0.01(\pm0.00)X_4$	0.05	0.0197
		Summer	$\text{Conc} = 3.9(\pm0.6) - 0.1(\pm0.03)X_2 - 0.05(\pm0.01)X_3 - 0.02(\pm0.00)X_4$	0.09	0.0024
		Fall	$\text{Conc} = 1.1(\pm0.20) - 0.03(\pm0.01)X_3 + 0.01(\pm0.00)X_4$	0.05	0.0086
		Winter	NS		
	RE1	Spring	$\text{Conc} = 1.4(\pm0.08) - 0.1(\pm0.02)X_2 - 0.02(\pm0.00)X_3$	0.15	0.0184
		Summer	$\text{Conc} = 0.5(\pm0.03) + 0.01(\pm0.00)X_1 - 0.06(\pm0.01)X_2$	0.12	0.0001
		Fall	$\text{Conc} = 1.3(\pm0.1) - 0.2(\pm0.02)X_2 - 0.04(\pm0.00)X_3$	0.41	0.0208
		Winter	$\text{Conc} = 1.5(\pm0.07) - 0.01(\pm0.01)X_1 - 0.2(\pm0.02)X_2 + 0.03(\pm0.01)X_3 - 0.01(\pm0.01)X_5$	0.27	0.0432
NO_2	RO1	Spring	$\text{Conc} = 75.0(\pm4.6) - 0.3(\pm0.1)X_1 - 4.3(\pm0.6)X_2 + 0.6(\pm0.1)X_3 - 0.4(\pm0.06)X_4$	0.25	0.0003
		Summer	$\text{Conc} = 93.2(\pm10.7) - 0.5(\pm0.2)X_1 - 2.8(\pm0.8)X_2 - 0.6(\pm0.1)X_4$	0.11	0.0088
		Fall	$\text{Conc} = 63.1(\pm7.3) - 6.9(\pm1.00)X_2$	0.11	0.0090
		Winter	$\text{Conc} = 74.1(\pm4.0) - 6.7(\pm0.7)X_2 - 0.2(\pm0.06)X_4$	0.28	0.0002
	RO2	Spring	$\text{Conc} = 3.5(\pm1.4)X_2 + 1.2(\pm0.3)X_3$	0.09	0.0393
		Summer	$\text{Conc} = 163(\pm9.8) - 5.1(\pm0.7)X_2 - 2.3(\pm0.3)X_3 - 0.9(\pm0.08)X_4 + 0.08(\pm0.04)X_5$	0.35	0.0001
		Fall	$\text{Conc} = 36.3(\pm5.6) + 0.5(\pm0.2)X_1 - 5.7(\pm0.7)X_2 - 0.5(\pm0.2)X_3 + 0.2(\pm0.08)X_4$	0.19	0.0032
		Winter	$\text{Conc} = 39.5(\pm2.5) - 3.8(\pm0.6)X_2 + 0.8(\pm0.2)X_3$	0.18	0.0003
	RE1	Spring	$\text{Conc} = 68.2(\pm2.5) - 0.4(\pm0.08)X_1 - 6.4(\pm0.4)X_2 - 0.2(\pm0.08)X_3 - 0.3(0.03)X_4$	0.48	0.0001
		Summer	$\text{Conc} = 62.5(\pm4.4) - 0.3(\pm0.08)X_1 - 5.1(\pm0.3)X_2 - 0.4(\pm0.05)X_4$	0.41	0.0001
		Fall	$\text{Conc} = 70.8(\pm3.9) - 0.4(\pm0.1)X_1 - 8.6(\pm0.6)X_2 - 0.3(\pm0.04)X_4$	0.32	0.0001
		Winter	$\text{Conc} = 47.8(\pm0.9) - 6.5(\pm0.3)X_2$	0.43	0.0001

Table 4 (*continued*)

Composition	Site	Season	Regression equation	R^2	P
Panel B					
O_3	RO1	Spring	Conc = 6.7(±2.3) + 0.2(±0.07)X_1 + 1.3(±0.3)X_2 + 0.40(±0.07)X_3 − 0.08(±0.03)X_4	0.27	0.0030
		Summer	Conc = 26.2(±4.9) + 0.4(±0.09)X_1 − 0.3(±0.05)X_4	0.40	0.0001
		Fall	Conc = 0.4X_1(±0.08) + 1.1(±0.3)X_2 + 0.3(±0.06)X_3	0.30	0.0003
		Winter	Conc = 0.4X_1(±0.06) + 1.7(±0.2)X_2 + 0.1(±0.06)X_5	0.36	0.0003
	RO2	Spring	Conc = 4.4(±1.5) − 0.3(±0.05)X_1 + 1.2(±0.4)X_2 − 0.2(±0.07)X_3	0.22	0.0036
		Summer	Conc = 24.0(±6.0) − 0.1(±0.06)X_4	0.09	0.0384
		Fall	Conc = 9.8(±1.6) + 2.1(±0.3)X_2 + 0.5(±0.04)X_3 − 0.1(±0.02)X_4	0.33	0.0001
		Winter	Conc = 7.5(±2.0) + 1.6(±0.2)X_2	0.25	0.0002
	RE1	Spring	Conc = 0.4(±0.06)X_1 + 3.8(±0.4)X_2 + 0.5(±0.07)X_3	0.29	0.0001
		Summer	Conc = 51.7(±5.9) + 0.5(±0.09)X_1 + 0.9(±0.4)X_2 − 0.8(±0.1)X_3 − 0.3(±0.05)X_4	0.39	0.0001
		Fall	Conc = 0.3(±0.09)X_1 + 3.1(±0.3)X_2 + 0.4(±0.07)X_3 − 0.07(±0.04)X_4 + 0.06(±0.02)X_5	0.36	0.0001
		Winter	Conc = 0.4(±0.09)X_1 + (±0.2)3.5X_2 + 0.2(±0.07)X_3 + 0.2(±0.08)X_5	0.49	0.0001
SO_2	RO1	Spring	Conc = 22.2(±1.7) − 0.2(±0.04)X_1 − 1.4(±0.2)X_2 − 0.09(±0.02)X_4	0.14	0.0001
		Summer	Conc = 18.4(±2.0) − 0.2(±0.04)X_1 − 0.1(±0.02)X_4	0.10	0.0001
		Fall	Conc = 35.5(±1.8) − 0.5(±0.05)X_1 − 2.5(±0.3)X_2 − 0.2(±0.02)X_4 − 0.04(±0.02)X_5	0.34	0.0001
		Winter	Conc = 27.7(±2.9) − 0.5(±0.1)X_1 − 1.8(±0.3)X_2 + 0.3(±0.1)X_3 − 0.4(±0.1)X_5	0.18	0.0181
	RO2	Spring	Conc = 25.5(±2.2) − 2.5(±0.4)X_2 − 0.09(±0.03)X_4	0.18	0.0036
		Summer	Conc = 18.7(±5.1) − 0.2(±0.08)X_1 − 1.3(±0.3)X_2 − 0.1(±0.1)X_4 − 0.04(±0.02)X_5	0.10	0.0322
		Fall	Conc = 43.0(±4.1) − 0.5(±0.1)X_1 − 2.5(±0.6)X_2 − 0.3(±0.05)X_4	0.13	0.0001
		Winter	Conc = 36.1(±5.5) − 1.2(±0.3)X_1 + 0.9(±0.2)X_3 − 0.2(±0.07)X_4 − 0.8(±0.3)X_5	0.11	0.0159
	RE1	Spring	Conc = 13.9(±1.1) − 0.08(±0.03)X_1 − 1.1(±0.2)X_2 − 0.06(±0.01)X_4	0.12	0.0091
		Summer	Conc = 4.3(±1.7) − 0.5(±0.1)X_2 + 0.1(±0.05)X_3	0.06	0.0333
		Fall	Conc = 18.9(±1.2) − 0.2(±0.04)X_1 − 1.2(±0.2)X_2 − 0.1(±0.02)X_4 − 0.02(±0.01)X_5	0.31	0.0001
		Winter	Conc = 15.4(±0.7) − 0.3(±0.05)X_1 − 1.1(±0.2)X_2 − 0.1(±0.06)X_3 − 0.2(±0.06)X_5	0.17	0.0252

[a] X_1, solar radiation (MJ/m^2); X_2, wind speed (m/s); X_3, air temperature (°C); X_4, relative humidity; X_5, precipitation (mm); NS, not statistically significant.

meteorological variables included in these equations varied according to the pollutant, monitoring station, or season. For O_3, all the regression equations were statistically significant. One notable result is that, in contrast to the roadside data, the regression equations obtained for the RE1 (residential monitoring station) O_3 data consistently included both X_1 (solar radiation) and X_3 (air temperature) for all seasons. This residential result is consistent with previous O_3 studies (Broomfield et al., 1996; Aneja et al., 2000; Krupa et al., 2003), which reported that both solar radiation and temperature are important meteorological variables for urban ambient levels. The R^2 values for the roadside stations were mostly lower than those for the residential station: the R^2 values of O_3 ranged from 0.09 to 0.40 for the roadside stations and from 0.29 to 0.49 for the residential station. It should be noted that R^2 represents the fraction of the variance in the air pollution levels due to the variability of the correlated meteorological parameters. Accordingly, the O_3 levels in the residential area were more influenced by meteorological conditions than the levels in the roadside areas. However, other confounding factors, such as long range transport, movement of synoptic and mesoscale weather systems, plus local production and regional transport of precursors (VOC and NO_x) and the associated photochemistry, would also have had an influence on the O_3 levels at the roadside and residential sites.

Although many previous efforts to relate air pollution data to meteorological variables have focused on O_3 (Broomfield et al., 1996; Aneja et al., 2000; Krupa et al., 2003), the present study also examined the relationships of the other target standard air pollutants (CO, NO_2, and SO_2). Most regression equations were statistically significant (Table 4(Panels A and B)). The R^2 values of these compounds were comparable to or somewhat smaller than those of O_3. For NO_2, the R^2 values ranged from 0.09 to 0.35 for the roadside stations and from 0.32 to 0.48 for the residential station, and for SO_2, from 0.10 to 0.34 for the roadside stations and from 0.06 to 0.31 for the residential station. Lastly, the R^2 values of CO ranged from 0.05 to 0.17 for the roadside stations and from 0.15 to 0.41 for the residential station. If validated by a further study, the regression

equations could be employed to partially predict air pollution levels based on analyzing meteorological data.

4. Conclusions

A roadside data were analyzed to provide baseline data for exploring associations between environmental exposure to four gaseous pollutants and health effects on residents living near roadways. The yearly roadside concentrations of CO and SO_2 showed a well-defined decreasing trend, whereas those of NO_2 and O_3 exhibited the reverse trend, suggesting that attention should be given to the NO_2 and O_3 exposure of residents living near roadways. In most cases, the diurnal trends of the roadside concentrations were well-defined for all seasons. The Sunday roadside concentrations of CO, NO_2, and SO_2 were similar to or somewhat lower than the weekday concentrations. Conversely, for O_3, the Sunday roadside concentrations were similar to or somewhat higher than the weekday concentrations. The monthly averages of O_3 concentrations exhibited the reverse seasonal variation to the other target compounds. It is also suggested that for O_3, the 8-h standard is more stringent than the 1-h standard, while for NO_2 and SO_2, the 1-h standard is more stringent than the 24-h standard. The multiple regression analyses confirmed that meteorological parameters are important for roadside pollution levels as well as residential pollution levels.

Acknowledgement

We would like to thank an anonymous reviewer of our manuscript for his/her great suggestions and other valuable comments. This work was partially supported by the Kyungpook National University Research Grant.

References

Alonso, L., Gangoiti, G., Navazo, M., Millán, M., Mantilla, E., 2000. Transport of tropospheric ozone over the Bay of Biscay and the eastern Catabrian coast of Spain. Journal of Applied Meteorology 39, 475–486.

Aneja, V.P., Adams, A.A., Arya, S.P., 2000. An observational based analysis of ozone trends and production for urban areas in North Carolina. Chemosphere—Global Change Science 2, 157–165.

Blanchard, C.L., Tanenbaum, S.J., 2003. Differences between weekday and weekend air pollutant levels in Southern California. Journal of the Air and Waste Management Association 53, 816–828.

Broomfield, P., Royle, J.A., Steinberg, L.J., Yang, Q., 1996. Accounting for meteorological effects in measuring urban ozone levels and trends. Atmospheric Environment 17, 3067–3077.

Bruetsch, R.I., 1981. Carbon monoxide and non-FTP ambient temperature. Office of Mobile Source Air Pollution Control, US Environmental Protection Agency, Ann Arbor, MI.

Chinkin, L.R., Coe, D.L., Funk, T.H., Hafner, H.R., Roberts, P.T., Ryan, A.R., Lawson, D.R., 2003. Weekday versus weekend activity patterns for ozone precursor emissions in California's South Coast Air Basin. Journal of the Air and Waste Management Association 53, 829–843.

Davidson, A., 1993. Update on ozone trend in California's South Coast Air Basin. Journal of the Air and Waste Management Association 43, 226–227.

Derwent, R.G., Middleton, D.R., Field, R.A., Goldstone, M.E., Lester, J.N., Perry, R., 1995. Analysis and interpretation of air quality data from an urban roadside location in central London over the period from July 1991 to July 1992. Atmospheric Environment 29, 923–946.

Draper, C., Smith, H., 1981. Applied Regression Analysis, second ed. John Wiely & Sons Inc., New York.

Edwards, J., Walters, S., Griffiths, R., 1994. Hospital admissions for asthma in pre-school children: relationship to major roads in Birmingham, United Kingdom. Archives of Environmental Health 49, 223–227.

Feister, U., Balzer, K., 1991. Surface ozone and meteorological predictors on a subregional scale. Atmospheric Environment A 25, 1781–1790.

Fuentes, J.D., Dann, T.F., 1994. Ground-level ozone in eastern Canada: seasonal variations, trends, and occurrences of high concentrations. Journal of the Air and Waste Management Association 44, 1019–1026.

Fujita, E.M., Croes, B.E., Bennett, C.L., Lawson, D.R., Lurmann, F.W., Main, H.H., 1992. Comparison of emission inventory and ambient concentration ratios of CO, NMOG, and NO_x in California's south coast air basin. Journal of the Air and Waste Management Association 42, 264–276.

Fujita, E.M., Stockwell, W.R., Campbell, D.E., Keislar, R.E., Lawson, D.R., 2003. Evolution of the magnitude and spatial extent of the weekend ozone effects in California's South Coast Air Basin, 1981–2000. Journal of the Air and Waste Management Association 53, 802–815.

Ghim, Y.S., Chang, Y.S., 2000. Characteristics of ground-level ozone distribution in Korea for the period of 1990–1995. Journal of Geophysical Research 105, 8877–8890.

Ghim, Y.S., Oh, H.S., Chang, Y.S., 2001. Meteorological effects on the evolution of high ozone episodes in the greater Seoul area. Journal of the Air and Waste Management Association 51, 185–202.

Heuss, J.M., Kahlbaum, D.F., Wolff, G.T., 2003. Weekday/weekend ozone differences: What can we learn from them? Journal of the Air and Waste Management Association 53, 772–788.

Huang, Y.L., Batterman, S., 2000. Residence location as a measure of environmental exposure: a review of air pollution epidemiology studies. Journal of Exposure Analysis and Environmental Epidemiology 10, 66–85.

Jo, W.K., Moon, K.C., 1999. Housewives' exposure to volatile organic compounds relative to proximity to roadside service stations. Atmospheric Environment 33, 2921–2928.

Jo, W.K., Yoon, I.H., Nam, C.W., 2000. Analysis of air pollution in two major Korean cities: trends, seasonal variations, daily 1-h maximum versus other hour-based

concentrations, and standard exceedances. Environmental Pollution 110, 11–18.

Judith, C.C., 2003. Introduction to special topic: Weekend and weekday differences in ozone levels. Journal of the Air and Waste Management Association 53, 771.

Kim, J., Han, B., Cho, S.Y., 2003. A numerical simulation of annual acid deposition amount in Korea. Atmospheric Environment 37, 1703–1713.

Kodama, Y., Arashidani, K., Tokui, N., Kawamoto, T., Matsuno, K., Kunugita, N., Minakawa, N., 2002. Environmental NO_2 concentration and exposure in daily life along main roads in Tokyo. Environmental Research 89, 236–244.

Krupa, S., Nosal, M., Ferdinand, J.A., Stevenson, R.E., Skelly, J.M., 2003. A multi-variate statistical model integrating passive sampler and meteorology data to predict the frequency distributions of hourly ambient ozone (O_3) concentrations. Environmental Pollution 124, 173–178.

Lee, H.S., Kang, C.M., Kang, B.W., Kim, H.K., 1999. Seasonal variations of acidic air pollutants in Seoul, South Korea. Atmospheric Environment 33, 3143–3152.

Logan, J.A., 1986. Tropospheric ozone: seasonal behavior, trends and anthropogenic influence. Journal of Geophysical Research 90, 10463–10482.

Mönkkönen, P., Uma, R., Srinivasam, D., Koponen, I.K., Lehtinen, K.E.J., Hämeri, K., Suresh, R., Sharma, V.P., Kulmala, M., 2004. Relationship and variations of aerosol number and PM_{10} mass concentrations in a highly polluted urban environment—New Delhi, India. Atmospheric Environment 38, 425–433.

National Research Council (NRC), 1991. Rethinking the ozone problem in urban and regional air pollution. National Academy Press, Washington, DC, pp. 50–65.

Pun, B.K., Seigneur, C., 2003. Day-of-week behavior of atmospheric ozone in three US cities. Journal of the Air and Waste Management Association 53, 789–801.

United States Environmental Protection Agency (USEPA), 1995. National air quality and emissions trends report, 1994. EPA/600/p-95/001/1aF. US Government Printing Office, Washington, DC.

van Wijnen, J.H., van der Zee, S.C., 1998. Traffic-related air pollutants: Exposure of road users and populations living near busy roads. Review of Environmental Health 13, 1–25.

Viana, M., Querol, X., Alastuey, A., Gangoiti, G., Menéndez, M., 2003. PM levels in the Basque Country (Northern Spain): Analysis of a 5-year data record and interpretation of seasonal variations. Atmospheric Environment 37, 2879–2891.

Wjst, M., Reitmeir, P., Dold, S., Wulff, A., Nicolai, T., von Loeffelholz, C.E., von Mutius, E., 1993. Road traffic and adverse effects on respiratory health in children. British Medicine Journal 307, 596–600.

Yang, J., Miller, D.R., 2002. Trends and variability of ground-level O_3 in Connecticut over the period 1981–1997. Journal of the Air and Waste Management Association 52, 1354–1361.