

Air Quality Impact of Biomass Co-Firing with Coal at a Power Plant in the Greater Houston Area

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Abstract

The Houston-Galveston-Brazoria (HGB) area of Texas is a moderate nonattainment region for ozone, and has a history of severe summer ozone episodes. W. A. Parish power plant (WAP) located in the greater Houston area is the largest coal and natural gas based electricity generating unit (EGU) in Texas. Forest residue is an abundant renewable resource, and can be used to offset coal usage at EGUs. This study evaluates the impact of co-firing 5%, 10%, and 15% (energy-basis) of forest residue at WAP on the air quality of the HGB area. Photochemical modeling with Comprehensive Air Quality Model with Extensions (CAMx) was conducted to investigate the air quality at three air quality monitoring sites (C696, C53, C556) in the HGB area, under two source scenarios (all-sources, point + biogenic sources). Significant reduction of SO₂ and O₃ was observed for 10% and 15% co-firing ratios at monitoring station (C696) close to WAP. The maximum reduction of ozone observed for 15% co-firing is 4.7% and 6.3% for all-sources and point + biogenic sources scenarios respectively. The reduction in other criteria air pollutants is not significant at all locations. The overall results from this study indicate that biomass co-firing at WAP would not lead to a significant reduction in ozone concentrations in the region during periods of peak ozone.

Keywords

Ozone, Houston, Photochemical Modeling, Biomass, Co-Firing

1. Introduction

The Houston-Galveston-Brazoria (HGB) region of Texas experienced high sum-

mer ozone episodes with formation rates rising as high as 200 ppbh⁻¹ in 2001, in contrast to ozone production rates around 40 ppbh⁻¹ in other cities of the US [1] [2]. The HGB area consists of eight counties: Galveston, Brazoria, Fort Bend, Chambers, Harris, Liberty, Montgomery and Waller, and is one of the 16 air quality regions administered by the Texas Commission on Environmental Quality (TCEQ) [3]. The HGB area was designated a moderate non-attainment area and was required to attain the 8-hour O₃ standard of 75 ppb which was set in 2008 [4]. A more stringent National Ambient Air Quality Standard (NAAQS) was mandated in 2015 and the eight-hour standard decreased to 70 ppb [4]. Ground level ozone formation depends on the complex reactions of volatile organic compounds (VOCs) and NO_x in the presence of solar radiation. The photochemical reactions that lead to the formation of ozone occur predominantly during hot, sunny summer days with high humidity levels [5] [6]. In this region, the formation of peroxy radicals ($\cdot\text{RO}_2$) which convert nitrogen monoxide (NO) back to nitrogen dioxide (NO₂) and reproduce hydroxyl radical ($\cdot\text{OH}$), sustains high net O₃ formation [7] [8]. Ground level ozone is largely a result of precursor emissions from anthropogenic sources such as petrochemical plants, fossil-fuel based electricity generating units (EGU), and mobile sources [9]. Power generation from fossil fuel emits a significant amount of NO_x, and emissions from EGUs can have a substantial impact on downwind air quality [10]. EGUs are the leading point source emitters of NO_x and SO₂ which are linked with the formation of photochemical ozone and acid deposition [11] [12] [13]. W. A. Parish (WAP) power plant is one of the largest coal-based power plants in the U.S. and is located in Fort Bend County of the HGB area. Rapid O₃ formation occurs when the anthropogenic VOC emissions from the Houston Ship Channel region combine with NO_x emissions from EGUs and mobile sources in the HGB area [9] [14]. WAP has eight units with a capacity of 3653 MW, with Units 1 - 4 fueled by natural gas and generating 1191 MW; Units 5 - 8 use coal and generate 2470 MW electricity. The concentration of aerosols and gas phase species from WAP is significantly higher compared to petrochemical industries in the region [15]. The Texas Air Quality Study II, on 19th September, 2006 tracked the plume from WAP and estimated the ozone production efficiency (OPE) as 4.4 at the plume age of 0.6 h, indicating that conditions are suitable for rapid O₃ formation when emissions from WAP combine with VOC emissions from the ship channel region [14].

An effective way to reduce emissions of criteria air pollutants from EGUs is to supplement coal usage with renewable energy sources such as, biomass, wind and solar [16] [17] [18] [19]. One of the renewable energy resources that are available locally in Texas, for power generation is forest residue [20]. Logging residue, the unused portions of harvested trees left in the woods, is potentially available for co-firing and includes tops, limbs, and un-utilized cull trees. Stumps are not feasible for co-firing, due to the cost of obtaining stump biomass being prohibitively high [20]. Currently, logging residues are either burned or

left in open fields by forestland owners as markets for logging residues are non-existent [21]. Biomass can be converted into electricity by direct combustion, co-firing with coal, gasification, and pyrolysis. Substituting biomass for a portion of coal in an existing coal power plant, termed co-firing has economic and environmental advantages [17]. Biomass can be co-fired at 10% - 25% (mass basis) without significant impacts to heat release characteristics of boilers [22] [23]. Co-firing is an economically feasible option for introducing new biomass power generation in most existing power plants. Coal can be replaced up to 15% by biomass in an existing power plant with minor modifications, and this replacement ratio is found to be optimum for cost-effectiveness [24]. Local availability of sufficient quantity of low-cost biomass is the leading factor for economic advantages of electricity production. U.S. electricity from biomass has increased by around 18% in ten years from 2005 to 2015, and 1.6% of the total electricity is from biomass as of 2015 [25]. A recent study by Kommalapati *et al.*, (2018) reported the changes in life cycle environmental impacts, across 15 mid-point impact categories, due to co-firing of forest residue with coal in the HGB area [19]. This life cycle assessment (LCA) study estimates that co-firing of biomass at the WAP plant would reduce life cycle NO_x and VOC emissions by 11.6% and 7.7%, for a co-firing ratio of 15% (energy basis) [19]. The current study builds on the estimates for combustion stage emissions, provided by Kommalapati *et al.*, (2018), and evaluates the impact on regional air quality of the HGB area, due to biomass co-firing at WAP plant. Photochemical modeling was conducted with the Comprehensive Air Quality Model with Extensions (CAMx), and results were visualized with Visualization Environment for Rich Data Interpretation (VERDI) tool. CAMx is a Eulerian photochemical dispersion model that simulates the emissions, dispersion, chemical reaction, and removal of pollutants by marching the Eulerian continuity equation forward in time for each chemical species on a system of nested three-dimensional grids [26]. The objective of the current study is to evaluate the impact of biomass co-firing at WAP power plant at the ratios of 5%, 10% and 15% (energy-basis), on the air quality of the HGB region during the June 2012 ozone episode.

2. Methodology

2.1. Emissions

Base-case emissions for combustion stage at the WAP plant, were obtained from TCEQ's Airs Facility Subsystem (AFS) files for modeling 2012 ozone episode. The average emissions for the four coal units of WAP plant, during June 2012 are described in **Table 1**, as emission rate in tonnes/hr (t/hr). Estimates for emissions reduction at the combustion stage were obtained from Kommalapati *et al.*, (2018), and are summarized in **Table 2** [19] [27]. Biomass co-firing is considered only in the coal units of WAP, which were Units 5 - 8, and were controlled by Selective Catalytic Reduction for NO_x .

Table 1. Average base-case emissions of air pollutants from combustion stage of coal units.

Pollutant	Measure	Unit-5 (650 MW)	Unit-6 (650 MW)	Unit-7 (560 MW)	Unit-8 (610 MW)
NO _x	(t/hr)	0.1818	0.1631	0.0982	0.1356
VOC	(kg/MWh)	0.2797	0.2508	0.1753	0.2224
CO	(t/hr)	0.0037	0.0044	0.0042	0.0049
PM _{2.5}	(kg/MWh)	0.0057	0.0068	0.0075	0.0081
SO ₂	(t/hr)	0.1256	0.0714	0.1401	0.2910

Table 2. Percentage reduction in combustion stage emissions due to co-firing [19].

Co-firing ratio	NO _x	VOC	CO	PM _{2.5}	SO ₂
5%	5.23%	2.80%	0.57%	2.94%	3.84%
10%	10.3%	5.60%	3.44%	7.44%	6.89%
15%	15.1%	8.40%	6.31%	11.9%	9.94%

2.2. Emissions Processing

Emission Processing System 3 (EPS3) consists of a series of FORTRAN modules that perform intensive data manipulations, producing an emissions inventory for photochemical modeling [26]. EPS3 was used to process the Airs Facility Subsystem (AFS) files publically available at the Texas Commission on Environmental Quality website [28]. This study used PREPNT, SPCEMS, TMPRL, PSTPNT, PIGEMS, GRDEM, MRGUAM modules to process data. The output from LCA study for combustion stage was used to modify only point sources file related to EGUs in the HGB area. All other emission files (Biogenic, area, non-road, offroad, oil and gas, onroad, and emissions from Mexico and Canada) are retained intact and used in final module (PIGEMS and MRGUAM) of EPS3 for merging.

2.3. Model and Site Selection

CAMx version 6.31 with improved halogen chemistry was used to predict air pollutant concentration in the HGB area. CAMx uses the 36 km master grid as coarsest grid and then moves to the finer grids of 12 km and 4 km. Lambert Conformal Conic map projection is used in Texas ozone modeling domains, where first true latitude, second true latitude, and central longitude are 33°N, 45°N and 97°W respectively. The modeling domain for this study includes emission sources in Canada, the U.S., and Mexico. The coarse domain covers the area under Easting (−2735, 2592) and Northing (−2088, 1944) coordinate, also the grid areas are divided into 148 × 112 cells. The tx_12 km domain has 149 × 110 numbers of cells and the tx_4 km domain is divided into 191 × 218 cells. The five major data classes needed as input for CAMx modeling are emissions, meteorology, photolysis, geographic, air quality. In this study, four simulations

were completed for each day, including a base case (no co-firing), a 5% co-firing case, a 10% co-firing case, and a 15% co-firing case. Each case has two scenarios: all-sources (biogenic, point, area, mobile), and point + biogenic sources. In all the cases, boundary condition, top concentration, meteorology, land use, photo-lysis are obtained from TCEQs *Texas Air Quality Modeling Files and Information (2012 Episodes)*, except low-level emission and elevated-level emission files [29]. The CAMx output is in binary format, and VERDI software is used to visualize these results. Air quality impact analysis is considered at three air monitoring stations to compare the effect of co-firing. The details of the three selected sites are provided in **Table 3** and **Figure 1**. University of Houston, Sugarland (C696) monitoring location, is selected as it is the closest monitoring station (11.02 km) to the WAP plant. Houston Bayland Park (C53) located at Harris County, and La Porte Sylvan Beach (C556); were selected due to high population density at Bayland Park and proximity to Houston Ship Channel for C556 respectively.

3. Results and Discussion

3.1. Nitrogen Oxides (NO_x)

The maximum concentration of NO_x in June 2012 was observed at 7:00 hr on 21st June, and 11th June for both base and point + biogenic cases respectively. Co-firing of biomass does not affect NO_x concentration during the highest concentration hours. There are some reductions observed due to co-firing, which occur when the concentration is less than 7 ppb for the base case, as presented in **Figure 2**. There is no change of NO_x level for the 5% co-firing case, except at 14:00 hr on June 3rd, where 1 ppb reduction was observed from a 3 ppb base-case concentration. On June 18th (8:00 hr), the concentration reduced to 12 ppb from 13 ppb for the 15% co-firing case. The possible reasons for the insignificant effect due to biomass co-firing could be that WAP power plant is controlled for NO_x emissions by using selective catalytic reduction (SCR), and 90% emissions reduction was already observed from 1997 to 2010 [9]. In addition, 67% of NO_x in the HGB area came from mobile sources in 2014, and even in point source emissions, non-EGUs emitted an almost double amount of NO_x than EGUs in 2012 [30]. So, considering all-sources for CAMx modeling may have a lower effect on ambient concentrations at the monitoring stations (C696). In addition, meteorological conditions like wind speed and wind direction may dampen the effect of reduced stack emissions on changes in ambient concentrations [31].

Table 3. Selected Monitoring Sites.

Site	Location	Latitude	Longitude
Site 1	University of Houston Sugarland (C696)	29°34'27"N	95°38'59"W
Site 2	Houston Bayland Park (C53)	29°41'45"N	95°29'57"W
Site 3	La Porte Sylvan Beach (C556)	29°39'19"N	95°00'35"W

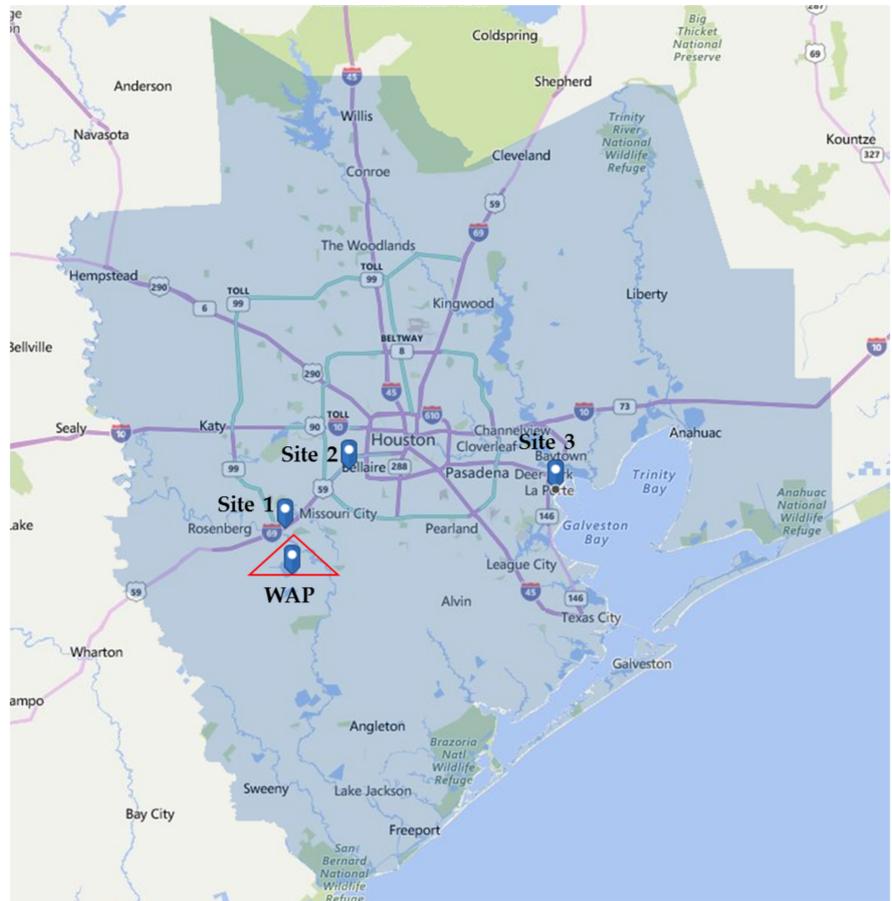


Figure 1. Location of air monitoring sites and WAP plant within the HGB area.

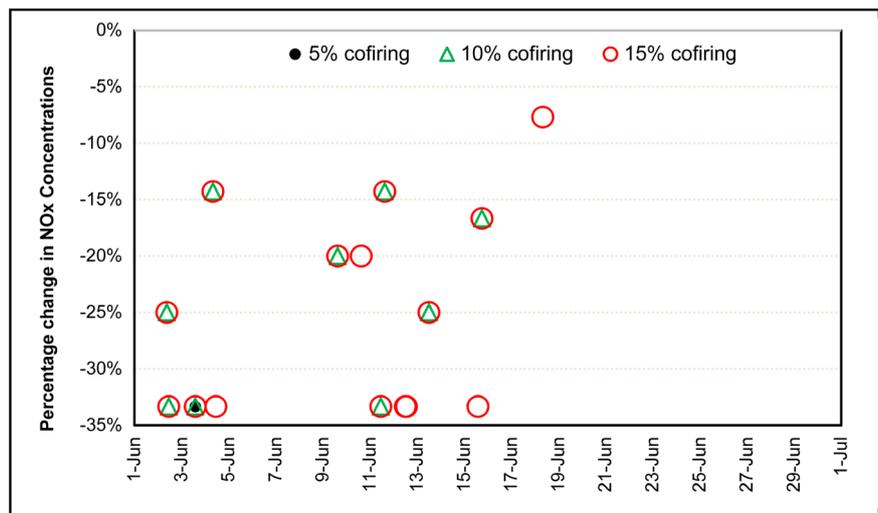


Figure 2. Reduction in NO_x concentrations at C696 station due to co-firing (all-sources).

There is no significant decrease in NO_x concentration due to co-firing, at the Houston Bayland Park monitoring station (C53). Only two and four hours out of the 720 hours for June 2012 showed a 1 ppb reduction for the all-sources and point + biogenic case scenarios respectively. Both of these reductions were dur-

ing low-concentration hours, reduction only observed for 10%, and 15% co-firing case. Onroad emissions of NO_x were 107.12 tons per day (tpd) in Harris County in 2012 which is 3 times more than EGUs emission of NO_x in the HGB area [3]. That means that C53 is dominated by mobile source emissions. Due to downwind wind pattern from the Gulf of Mexico, the NO_x emissions from a high stack of WAP can be transported to C53 location and influence the air quality depending on meteorological conditions of the day. Co-firing of biomass in WAP power plant does not have an effect on C556 station when all source considered. For point + biogenic sources, only three different hours shows reduction: The maximum reduction observed at 10:00 hr (6th June) which was 1 ppb reduction from 4 ppb (25%); 1 ppb reduction observed from base case 25 ppb for both 10% and 15% ratios at 8:00 on the same day; and 5th June at 9:00 hr showed 1 ppb reduction from 7 ppb in the 10% and 15% ratio cases. The dominant sources in 2012 for NO_x were On-road emissions in Harris County. Also, C556 located in the Houston Ship Channel (HSC) region where marine emissions have a higher impact than point sources from Fort Bend County. HSC is the largest complex of industrial sources of NO_x and VOC in HGB area [9]. The wind flow passes WAP power plant goes to Houston downtown mostly, so it is unlikely that wind carries NO_x to C556. But wind flow observed by Darby (2005) in higher ozone days in Houston often had offshore flow in the morning that switched to onshore flow in afternoon [32].

3.2. Volatile Organic Compounds (VOC)

This study added 14 different species (ALD₂, ALD_x, ETH, ETHA, ETOH, FORM, IOLE, ISOP, MEOH, OLE, PAR, TERP, TOL, and XYL) of VOCs in VERDI to get VOC emissions. There is no significant variation of VOCs due to co-firing in ambient concentrations at C696 station, for the all-sources scenario, as presented in **Figure 3**. The variation was only observed during lower concentration days, only for four hours in the whole month. The maximum reduction was observed on 14th June (6%), which is only 1 ppb reduction from a 16 ppb base-case concentration. Considering only point + biogenic sources at C696, only six hours out of 720 hours, showed a reduction. The maximum drop was at 11:00 hr on 12th June of 1 ppb reduction from a 9 ppb base-case, for both 10% and 15% co-firing. There is no reduction observed due to 5% co-firing, except at 13:00 hr on 2nd June. At station C53, only one hour in the whole month showed a reduction (1ppb) in both cases for 15% co-firing. Co-firing of biomass in WAP has no effect on C556 station through the month, for both source scenario. Major contributors of VOC emission in HGB are area sources. In 2014, VOC emissions contribution from the area, mobile, and point sources are 61%, 23%, and 16% respectively [33]. C53 located far from WAP in Harris County where mobile emissions much higher than Fort Bend County. The probable reason for negligible reduction may be a wind effect which carries the plume from WAP to C53 on this particular hour. C556 is dominated by VOC emission from Houston

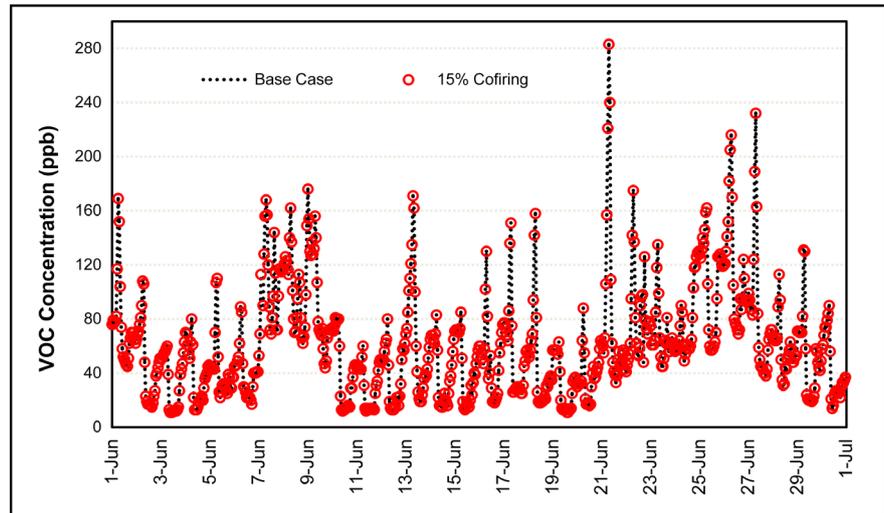


Figure 3. Reduction in NO_x concentrations at C696 station due to co-firing (all-sources).

Ship Channel. In 2006, the VOC emission from WA Parish power plant and HSC was 0.5 tpd and 58.7 tpd respectively [9]. Temperature and wind account for 66% of the variation of VOCs from point sources [34].

3.3. Ozone (O_3)

The maximum concentration during the June-2012 episode (105 ppb) was observed at 13:00 hr on 26th June at C696 station for all-sources. The main reason behind maximum ozone on 26th June is the high temperature and low cloud cover [35]. Biomass co-firing at WAP power plant has marginal reductions in ozone concentrations at the C696 location, and co-firing has effects on daily peak ozone concentrations as described in **Figure 4**. There are 23 different hours out of the 720 hours that show a reduction of ozone due to co-firing. The maximum percentage reduction (4.76%) was observed on 3rd June, but it is during the low ozone level. There are 7 hourly periods (1st June, 16:00 hr; 6th June 11:00 hr; 14th June, 11:00 hr; 15th June, 11:00 and 13:00 hr; 18th June, 12:00 hr; and 27th June, 13:00 hr) that co-firing resulted in reduction of daily maximum 1-hr ozone concentrations. 1 ppb ozone reduction from peak can prevent 2 - 3 deaths per year, indicating the significance of achieving marginal reduction [36]. Time series graph of concentrations for the only point and biogenic sources are presented in **Figure 5**. The maximum concentration for this sub-case (68 ppb) was observed at 13:00 hr on June 27th. Co-firing has a significant effect on air quality at C696, the maximum percentage (6.25%) reduction was observed at 11th June (20:00 hr) for 15% co-firing, ozone reduced to 15 ppb from 16 ppb. Reduction of ozone was also detected on higher concentration days. On June 1st (17:00 hr), 1 ppb ozone reduction for 15% co-firing (base case ozone 62 ppb) was observed, and 1 ppb decreases for both 10% and 15% co-firing cases was observed on 27th June (12:00 hr). The ozone formation of C696 station is both NO_x and VOC sensitive and is in a transitional state. So, the reduction in O_3 at C696 may occur

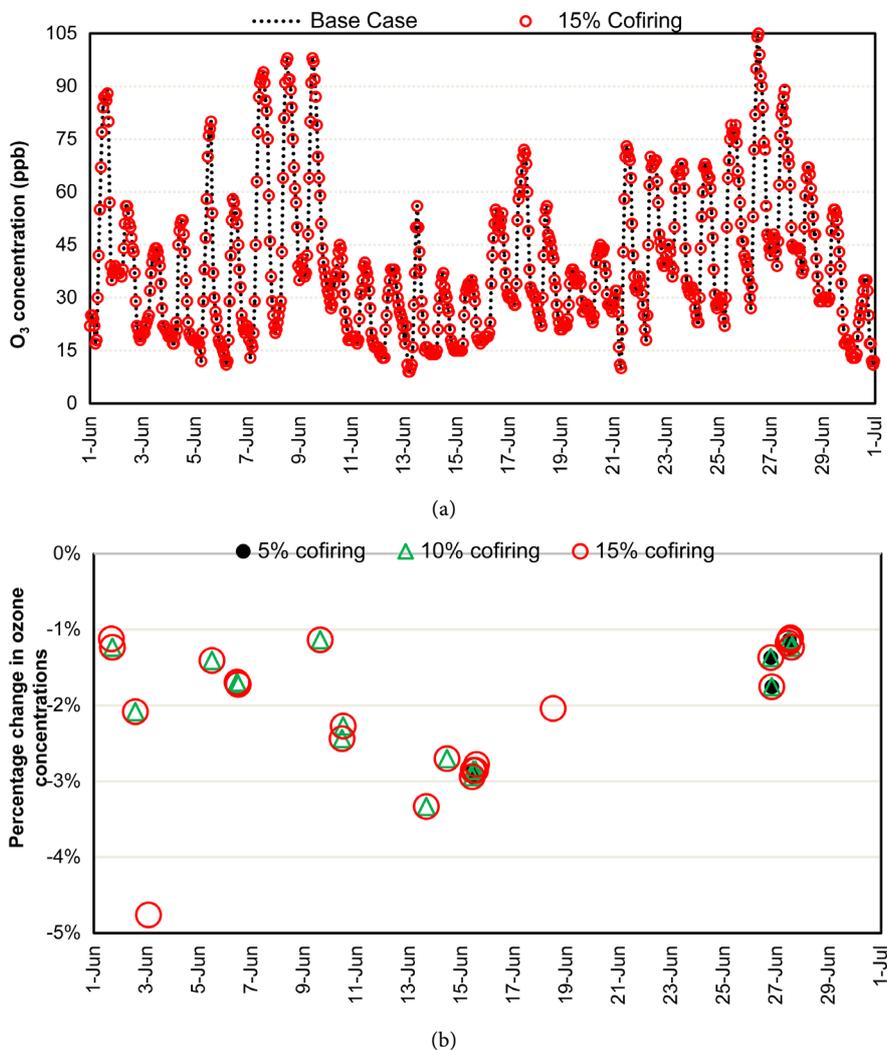


Figure 4. Effect of biomass co-firing on O₃ concentrations at the C696 site (all-sources), (a): O₃ concentrations for June 2012; (b): Percentage change due to co-firing.

due to the reduction of both NO_x and VOC from the power plant. A study reported that 50% reduction of NO_x from the HGB area has a huge impact on ozone in Houston and showed peak ozone reduced by 15% at air monitoring station located at the east of the HSC [37]. Also, previous studies found that ozone increased in areas where mobile and point sources dominate NO_x emissions and regions with lower NO titration. Lin *et al.*, (2005) observed that the peak hourly ozone decreases by 70 ppb and 128 ppv in Houston when NO_x and VOC emissions from Texas point sources were ignored [38]. Also, 3% ozone concentration difference occurred for 0.16% point source emission difference in HGB. Our study shows that a small but significant change of ozone at C696 station can be observed for very low reduction of VOC and NO_x from point sources [39]. A report by TCEQ, noted that the peak ozone on September 8, 1993 reduced 1ppb from 187 ppb in Houston due to complete removal of NO_x emission from WAP [40].

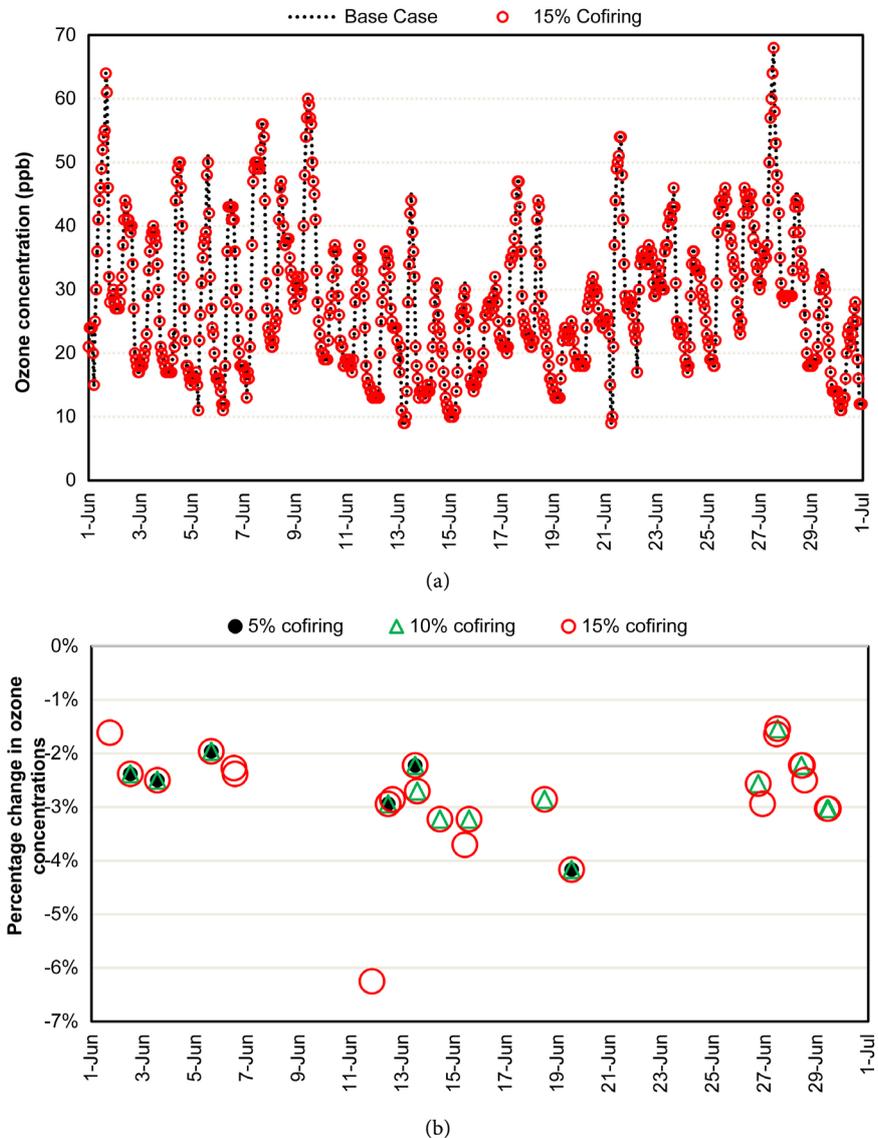


Figure 5. Effect of biomass co-firing on O_3 concentrations at the C696 site (point + biogenic sources), (a): O_3 concentrations for June 2012; (b): Percentage change due to co-firing.

From the CAMx results of O_3 concentrations at C53 monitoring station for the base case and co-firing cases, percentage change in hourly ozone was analyzed and presented in **Figure 6**. The daily maximum occurred mostly during 11:00 to 14:00 hr, and the maximum during the month of June was observed on 26th June (11:00 hr and 13:00 hr). Though there were some hours where ozone reduction was observed due to co-firing at WAP, overall reduction was insignificant at C53, for all-sources. The maximum concentration for the month of June (64 ppb) was observed at 15:00 hr of 9th June by considering the point and biogenic sources. The maximum percentage reduction (3.33%) occurred at 13th June (10:00 hr), the reduction is 1 ppb for 5%, 10%, and 15% co-firing ratios (base case concentration, 29 ppb). The changes due to 5% co-firing occurred

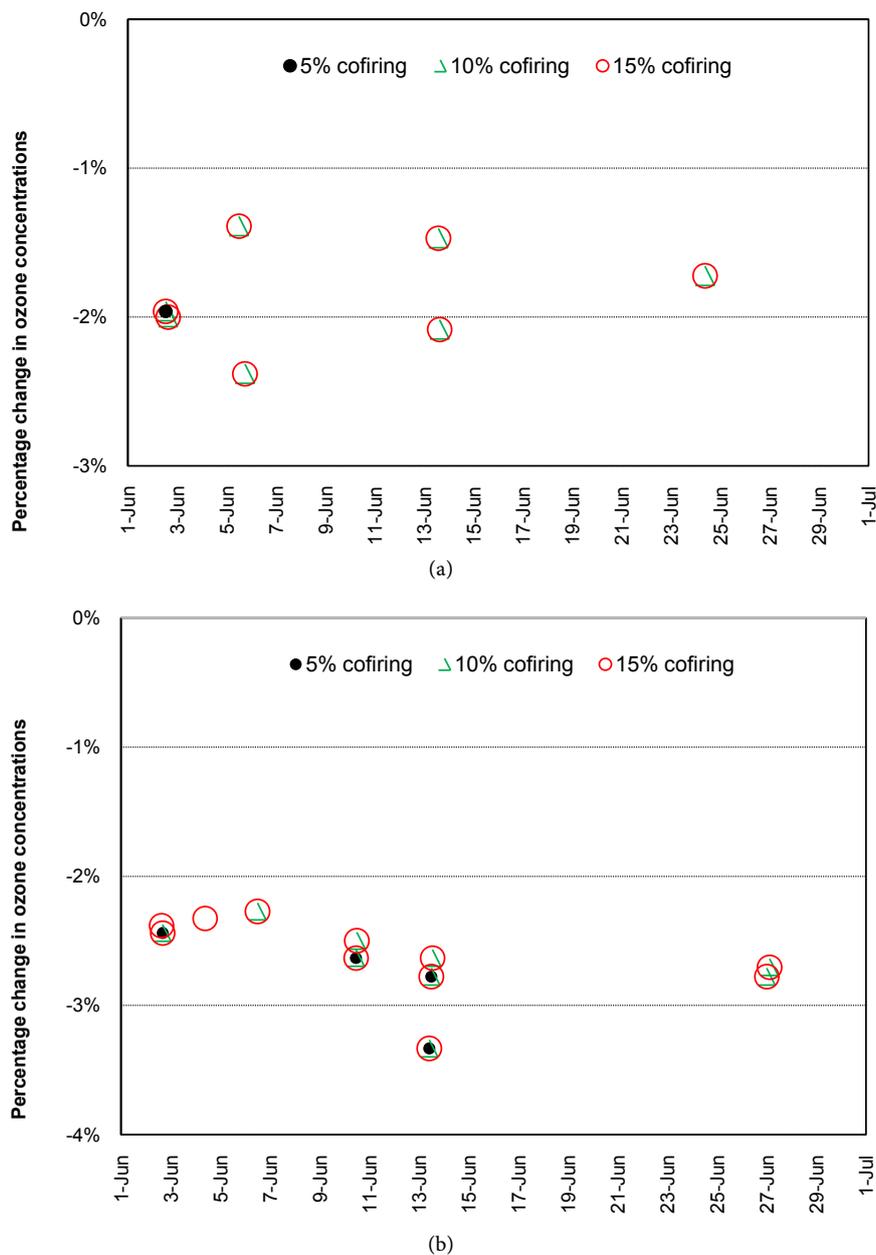


Figure 6. Percentage change in ozone concentrations at C53 in June 2012, due to co-firing; (a): all-sources; (b): point + biogenic sources.

several times in this case (2nd June, 10th June, and 13th June). Biomass co-firing has not affected ozone levels during the peak concentration hours. Houston Bayland Park is mostly sensitive to VOC. So, the reduction of ozone might occur due to the reduction of VOC. The ozone concentrations did not show any significant changes due to co-firing at C556 for all-sources case. Only one incident of reduction was observed at 13:00 hr on 2nd June where, 1 ppb ozone reduced due to 15% co-firing, from a base case of 66 ppb. This reduction occurred on a relatively high ozone day. Only three hours out of 720 hours for the month of June showed a reduction for the point + biogenic case. The maximum percen-

tage reduction (14%) occurred on June 1st (0:00 hr), where actually 1ppb reduction noted from 7 ppb (base case) for all three co-firing cases. The monitoring station that is located in the HSC area is mostly affected by VOCs. C556 has VOC sensitive ozone in most of the hours in June'12. High ozone concentrations in the HSC area are attributed to high concentrations of hydrocarbons [41]. So, the reduction of ozone precursors from WAP has a negligible effect at C556. One of the reasons for observing negligible effect at C556 is due to the westerly wind flow that carries plume from WAP at night and influences ozone only during subsequent days. But if the winds are very light, it creates stagnant conditions in the local area where the plume is generated. Also, nighttime transport of NO_x reduces the concentration of NO_x. WA Parish plume of NO_x losses 36% to 44% on average in overnight transport and the maximum reduction for 12h of transport in darkness was 73%, which lead to lower ozone formation due to Parish plume in downwind [42]. The reason for this was lack of enough NO_x at plume to titrate background ozone which led to rapid nighttime oxidation of NO₂ during downwind transport during that particular simulation period.

C556 air monitoring station is located in Harris County and nearby the Houston Ship Channel. Time series graph for ozone concentration at C556 is presented in **Figure 7**. When all-sources were considered, the maximum concentration for June month was observed at 15:00 hr on 26th June (99 ppb). The daily peak occurred during 12:00 to 14:00 hr on most of the days of the month, and June 20th the peak occurred at 23:00 hr. The diurnal variations followed the same trend as other locations, lower concentrations observed in the early morning. The ozone concentrations did not show any significant changes due to co-firing. Only one incident of reduction was observed at 13:00 hr on 2nd June where, 1ppb ozone reduced due to 15% co-firing, from a base case of 66ppb. For the point + biogenic sources scenario, the maximum concentration was observed on 26th June (9:00 hr) (72 ppb). The peak ozone in the first 10 days of the month was observed on 5th June and for the mid-10 days on 17th June, the peaks are 55 ppb and 47 ppb respectively. In mid-10 days the daily peaks of ozone concentrations are relatively lower than other parts of the month. The lower concentration occurred during midnight to morning in most of the days. There is no significant reduction of concentrations at this location due to co-firing. Only three hours out of 720 hours for the month of June show reduction. The maximum percentage reduction (14%) occurred on June 1st (0:00 hr), where actually 1ppb reduction noted from 7 ppb (base case) for all three co-firing cases. The other two hours where the reduction was observed were at 5th June 12:00 and 13:00 hr, where 1 ppb reduction observed from 55 ppb and 50 ppb respectively.

3.4. Sulfur Dioxide (SO₂)

Lower concentrations of SO₂ were observed during midnight to early morning throughout the month, except June 7. The peak SO₂ concentration (34 ppb) of June 2012 observed on 13th June, 11:00 hr, as shown in **Figure 8**. The daily

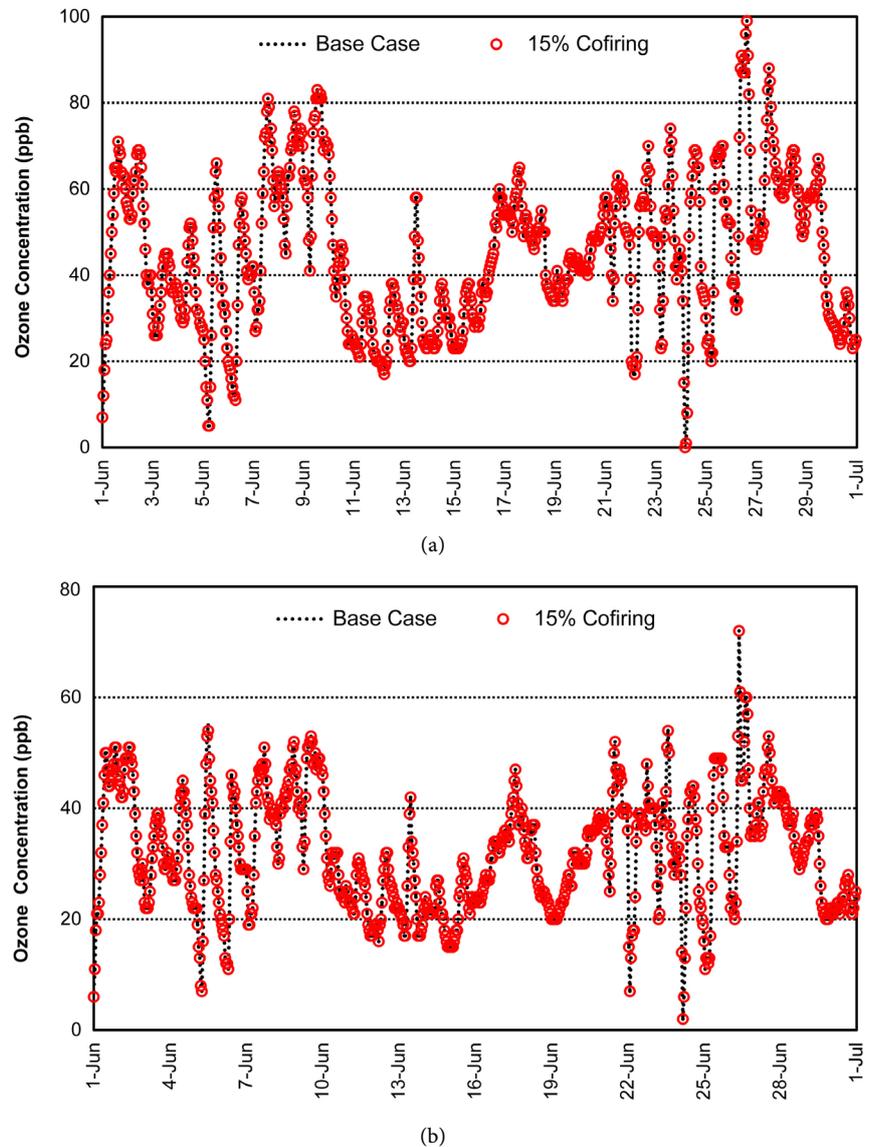


Figure 7. Effect of biomass co-firing on O₃ concentrations at C556 (a): all-sources; (b): point + biogenic sources.

maximum was observed in the mornings between 7:00 to 11:00 hr for most of the days. Ambient SO₂ concentrations at C696 air monitoring station were significantly affected due to co-firing of biomass with coal at the WAP power plant. The maximum reduction (25%) of SO₂ observed on 5th June in the first 10 days of month, but it is in lower concentration day. Sulfur dioxide concentrations also show significant drop during higher concentration days also. On June 4, 8:00 hr the peak SO₂ in first 10 days, due to co-firing of 5%, 10%, and 15% ratio the concentration reduces by 5%, 5%, and 10% respectively (**Figure 8(b)**). The peak for the whole month was observed at June 13th; concentrations show a gentle decrease in this day, which is 9% and 12% for 10% and 15% co-firing respectively. In higher concentration days, 5% co-firing does not have an impact mostly. Almost all of the higher concentration days observed a drop of concentration due

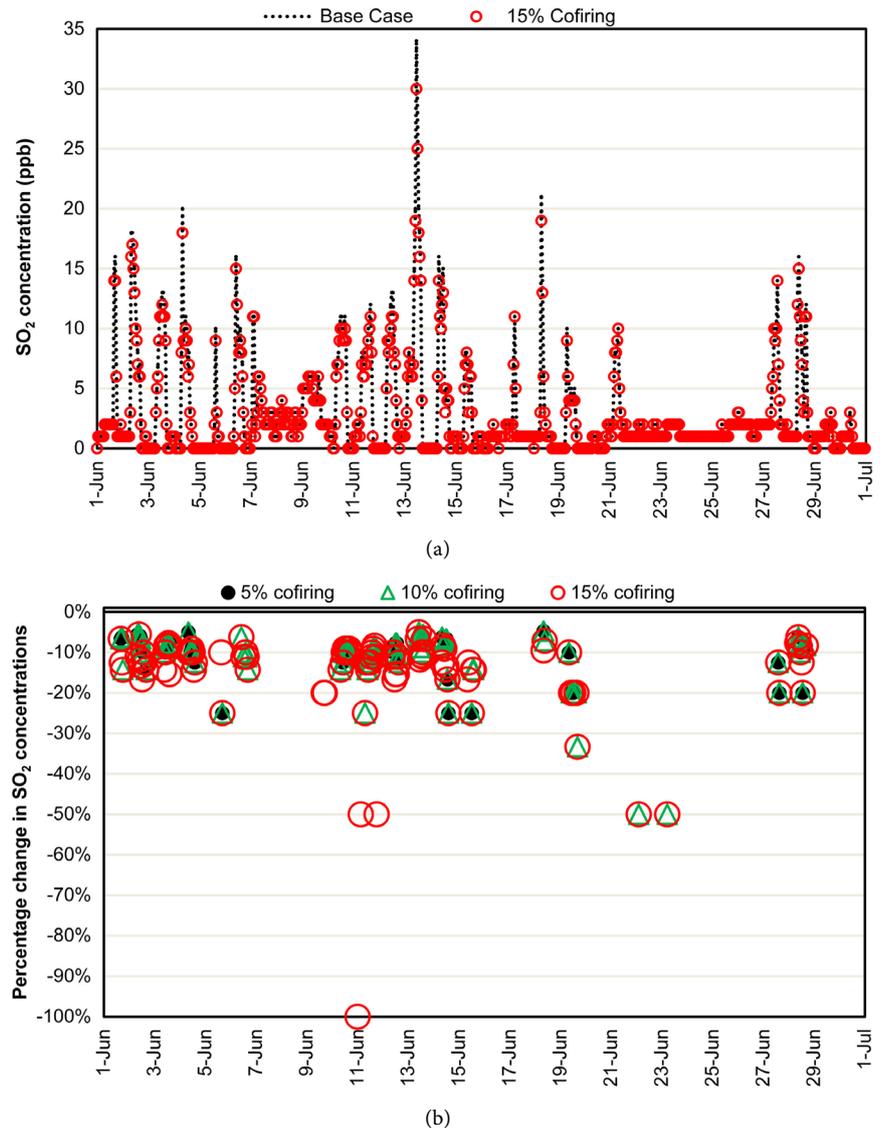


Figure 8. Effect of biomass co-firing on SO₂ concentrations at C696 (a): Time-series graph; (b): Percentage change due to co-firing.

to co-firing while lowest SO₂ concentration didn't show a frequent drop. For the point + biogenic sources, the maximum concentration (30 ppb) was observed at 11:00 hr on June 13th. It is the same time where peak occurred due to all source case also. The daily maximum observed for all days during daytime except June 7th where daily peak shows at 2:00 hr. The reduction of SO₂ concentration observed for almost all the high concentration hours with less number of hours counted that have a lower concentration. The reduction in peak concentration day (13th June) is 3 ppb (32 ppb to 29 ppb) for 15% co-firing, 2 ppb for 10%, and 1 ppb for 5% co-firing.

Fossil fuel combustion is the largest source of SO₂ emissions. Power plants emit more than 70% of SO₂ in Houston, other industrial facilities contribute around 20%. TCEQ studied 7 major point source emissions sources in HGB area

where WAP parish plant had highest (167.1 tpd) SO₂ emission in 2006 that was around two times greater than all other six sources [9]. Emissions from non-road sources in Fort Bend County were relatively lower (0.01 tpd) in 2012, and area sources emitted 0.28 tpd, which was not significant [43]. There are no significant sources of SO₂ in nearby WAP power plant. Since C696 is nearby WAP power plant, Power plant emission has a great impact on air monitoring station in both cases. The highest SO₂ days affected most because of WAP is probably responsible for high SO₂ in this station. Also, wind direction and speed plays a vital role for highest concentration days. A study by TCEQ shows that Croquet monitoring station located at 22.4 km north-northeast of Parish is heavily affected when the wind blows towards station from WAP [27]. There are a few reduction hours observed in C556 for all-sources case. The maximum reduction (13%) observed at 12:00 hr and 11:00 hr on 5th June and 6th June for 15% co-firing. The other time periods where the reduction was observed are at 7:00, 8:00, 9:00 hr on June 5th and 9:00 hr on June 6th, all these hours indicating a 1 ppb reduction for 10% and 15% co-firing. No reduction observed for 5% co-firing. The reasons behind insignificant reduction that C556 located at HSC where ship emissions have a dominant impact. For point + biogenic case, the peak SO₂ concentration observed at 0:00 hr on June 1st and the value is 23 ppb and 1 ppb reduction observed in that hour for 10% and 15% co-firing cases. There is no big drop in concentration overall except at 10:00 hr on 5th June where 1 ppb dropped from 4 ppb for both 10% and 15% co-firing ratio which is a 25% reduction.; at 7:00 hr, 8:00 hr in same day 1 ppb drop observed for same co-firing cases. Also, 1 ppb reduction observed at 9:00 and 10:00 hr on 6th June.

3.5. Carbon Monoxide (CO)

Figure 9 describes the percentage reductions observed in CO concentrations, due to co-firing considering all-sources. Co-firing has a negligible effect on CO concentrations at the C696 station. There is no change in emissions for the whole month, due to 5% co-firing, except on June 2 and 12 at 9:00 and 10:00 hr of 0.78% and 0.82% respectively. For 10% co-firing, there are changes in 16 hourly data points out of 720 hours, at a maximum reduction of 1.04% on 4th June. For the 15% co-firing case, a total of 27 hourly data points show a reduction, with a maximum reduction similar to 10% case. There is no reduction observed during the peak CO concentration hours. Any reductions which were only observed during lower concentration hours would not affect the CO levels in view of public health or welfare. A comparison of the base case and co-firing cases doesn't indicate a significant difference due to limiting the source categories to point + biogenic sources. The maximum reduction of 10.23%, on June 5th at 15:00 hr for 15% co-firing ratio, CO reduces from 88 ppb to 79 ppb at this particular hour. The maximum reduction of 2% CO, for 5% co-firing was observed on June 1, 21:00 hr. All of the reductions occurred when CO was less than 150 ppb. C696 is the closest air monitoring station to WAP, and Fort Bend

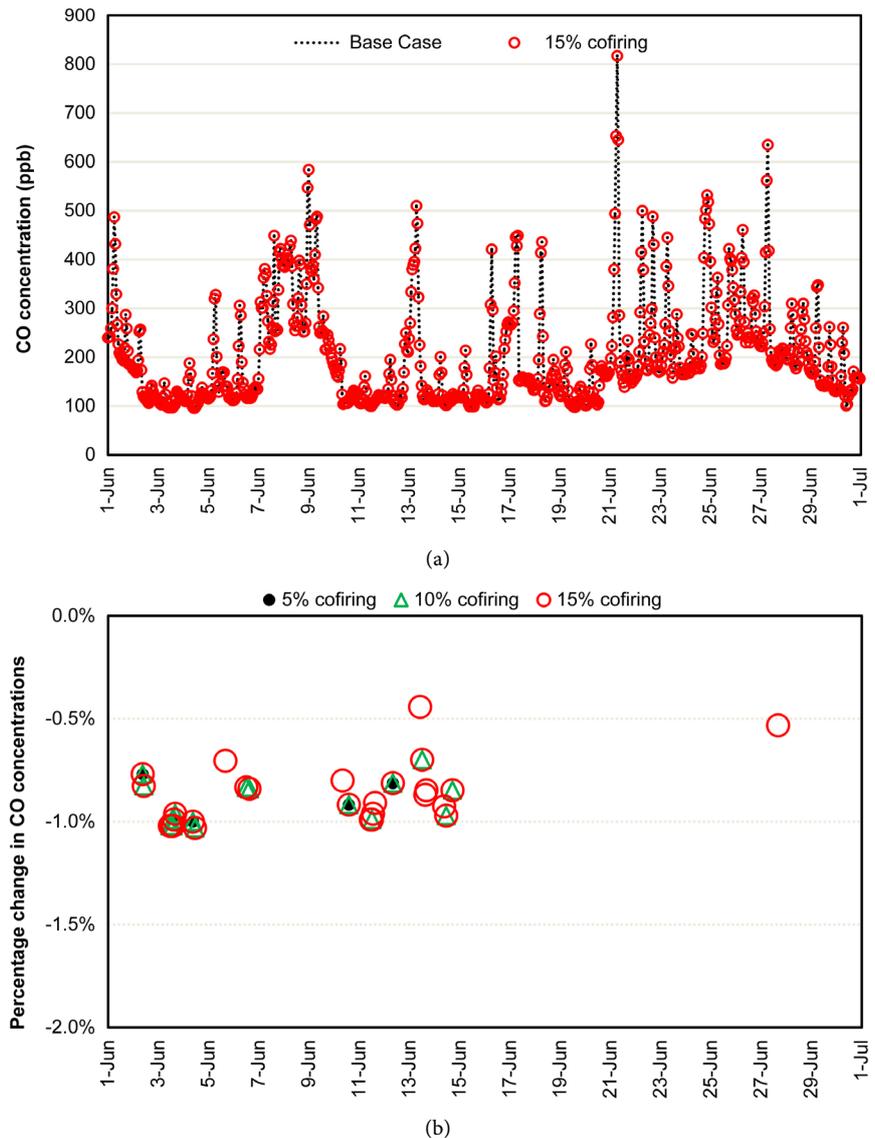


Figure 9. Effect of biomass co-firing on CO concentration at C696 (all-sources) [(a): CO concentration; (b): Percentage change].

County has lower point source emissions (WAP excluded), than Harris County. When all-sources were considered, modeling results suggest that reduction in CO levels would be significantly lower compared to the case where only point and biogenic sources are considered. This can be explained by the fact that mobile sources emit huge quantities of CO in composition to point sources, thereby magnifying any effect of co-firing at WAP on the CO levels observed at C696 station. Also, meteorological factors greatly influence pollutant concentrations, minimizing the effects of co-firing. The co-firing of biomass with coal in WAP does not have any notable effect on C53 and C556 air monitoring station in both cases. Only on 10th June 7:00 hr, a very small reduction of 1ppb from 503 ppb, was observed for 10% and 15% co-firing. C53 is located in highly populated area which is greatly affected by mobile sources rather than electricity generation fa-

cilities and is located 27.64 km far from WAP. Friedfeld *et al.*, (2002) reported that 90% of total CO came from mobile sources whereas only 8% is due to all stationary sources in HGB area for 1999 [44]. So, a small percentage of CO changes in WAP do not have a significant effect on C53 location. Area sources emit almost 10 times CO in Harris County than Fort Bend County [27]. C556, located in Harris County and non-road emissions of CO are around 13 times higher than Fort Bend County, where WAP is located. So, this small reduction of CO in WAP due to co-firing does not necessarily affect C556 air quality. **Table 4** summarizes the number of reductions in daily maximum 1 hr concentrations of all pollutants that were observed at the C696 monitoring station, during the month of June 2012.

4. Conclusion

This study evaluated the air quality impact of biomass co-firing with coal at three monitoring sites (C696, C53, C556) in the HGB area with CAMx simulations for the month of June 2012. Co-firing ratios of 5%, 10%, and 15% (energy-basis) of forest residue at WAP were considered, and combustion stage emissions were modified in the point source AFS file for EGUs, as input to CAMx simulations. Two source scenarios (all-sources, point + biogenic sources) were considered and significant reduction of SO₂ and O₃ was observed for the 10% and 15% co-firing ratios at the monitoring station (C696) closest to the WAP plant. The maximum reduction in ambient ozone concentrations was observed for 15% co-firing, around 4.7% and 6.3% for all-sources and point + biogenic sources scenarios respectively, on high concentration days. The ambient concentrations at monitoring stations far from WAP were not affected significantly by emissions reduction from co-firing. The maximum reduction observed for SO₂ at the C556 station was estimated to be 13% (all-source) and 25% (point + bio), for the 15% co-firing ratio, indicating the importance of WAP SO₂ control systems for the HGB region. Ozone sensitivity varies with location, and the influence of WAP to VOC levels is negligible at all stations. The results indicate that biomass co-firing at WAP plant would only impact ambient concentrations of ozone and SO₂ at the C696 station, which is the closest to the plant, and this reduction is more pronounced when considering (point + biogenic) sources

Table 4. Number of reductions in daily maximum 1-hr concentrations at C696 station, for June 2012.

Air Pollutant	All-sources	Point + Biogenic
SO ₂	17/30	19/30
NO _x	-	9/30
VOC	-	-
CO	-	1/30
O ₃	5/30	9/30

instead of all-sources. The overall results from this study indicate that biomass co-firing at WAP in isolation, would not lead to a significant reduction in ozone concentrations in the region, especially during periods of peak ozone levels. Biomass co-firing should be combined with emission controls for the area and mobile sources in order to observe any further significant improvement. Certain limitations of the current study, such as duration of CAMx simulations and non-inclusion of meteorological fluctuations should be considered while interpreting the results described in this study.

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Conflicts of Interest

The authors declare no conflicts of interest regarding the publication of this paper.

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Appendix-A

Comparison of modeled and observed concentrations: At the C696 station, the maximum observed hourly ozone concentrations of 109 ppb occurred on 26th June, 14:00 hr, followed by 94 ppb on 1st June, 16:00 hr. The corresponding CAMx modeled ozone was 99 ppb and 89 ppb respectively (Figure A-1). Bayl-and Park (C53) monitoring stations show a similar trend as C696, with observed maximums of 135 ppb on 1st June, 17:00 hr followed by 115 ppb at 26th June, 14:00 hr, the corresponding modeled ozone concentrations were 80 ppb and 104 ppb respectively (Figure A-1). In the case of C556 station, the observed maximum

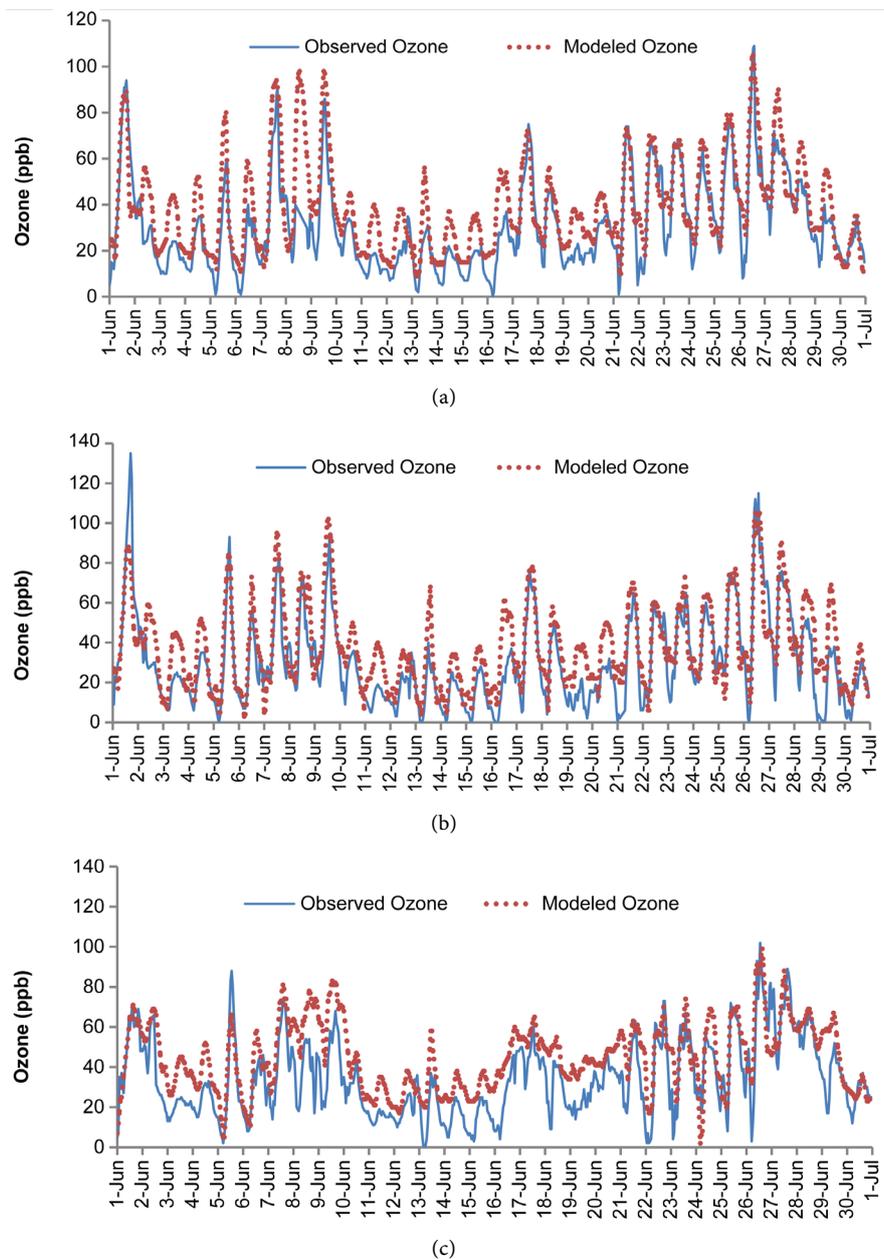


Figure A-1. Observed vs. modeled concentrations for comparison at (a) C696; (b) C53; (c) C556.

of ozone was 102 ppb on 26th June, 13:00 hr, while 87 ppb in CAMx simulation (**Figure A-1**). The peak ozone occurs in afternoon for all three cases and simulated ozone consistently underestimates during high ozone episodes. Low ozone concentrations occur during midnight to morning. Meteorological inputs are the dominating factors for major uncertainty in photochemical modeling [45] [46]. Wind field, temperature, and cloud cover are the determining factors among the meteorological variables [47]. Cloud cover is not fully resolved by air quality models, which causes variation in predicted and observed ozone [42]. The surface boundary layer of the atmosphere is very shallow during night time, which is poorly described by meteorological models. Local sources of NO_x emissions (e.g., nearby road) can exert a strong influence on ozone that is not replicated in a regional air quality model with 4km square grid cells [48]. This study also applied statistical parameters to evaluate the performance of CAMx modeled vs. observed ozone concentrations. Two statistical parameters were used in this study: Fractional bias (FB) and coefficient of determination (R^2). If FB value ranges -0.5 to $+0.5$, it can be inferred that the model performance is acceptable within 95% confidence interval [49]. R^2 indicates the proportion of variation in measured ozone concentrations that can be explained by modeled ozone and evaluate how close the data fit the regression line. The ideal value of R^2 is 1. **Table A-1** shows the FB and R^2 value for all three stations.

Table A-1. Observed vs. modeled concentrations for comparison at (a) C696; (b) C53; (c) C556.

Site name	Fractional bias	R^2 value
C696 UH of Sugarland	0.267	0.7844
C53 Houston Bayland Park	0.231	0.7465
C556 La Porte Sylvan Beach	0.273	0.6600