EMISSION INVENTORY OF NEPAL'S AIR POLLUTANTS FOR EFFECTIVE AIR QUALITY MANAGEMENT

A THESIS SUBMITTED TO THE
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FOR THE AWARD OF
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BY

BHUPENDRA DAS
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RECOMMENDATION

This is to recommend that **BHUPENDRA DAS** has carried out research entitled **"EMISSION INVENTORY OF NEPAL'S AIR POLLUTANTS FOR EFFECTIVE AIR QUALITY MANAGEMENT"** for the award of Doctor of Philosophy (Ph.D.) in **ENVIRONMENTAL SCIENCE** under our supervision. To our knowledge, this work has not been submitted for any other degree. He has fulfilled all the requirements laid down by the Institute of Science and Technology (IOST), Tribhuvan University, Kirtipur for the submission of the thesis for the award of Ph.D. degree.

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LETTER OF APPROVAL

On the recommendation of **Prof. Dr Rejina Maskey Byanju, Dr Prakash V. Bhave and Dr Siva PravenPuppala**, this Ph. D. thesis submitted by **Bhupendra Das**, entitled **"EMISSION INVENTORY OF NEPAL'S AIR POLLUTANTS FOR EFFECTIVE AIR QUALITY MANAGEMENT"** is forwarded by Central Department Research Committee (CDRC) to the Dean, IOST, T.U.

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DECLARATION

Thesis entitled “**EMISSION INVENTORY OF NEPAL’S AIR POLLUTANTS FOR EFFECTIVE AIR QUALITY MANAGEMENT**” which is being submitted to the Central Department of Environmental Science, Institute of Science and Technology (IOST), Tribhuvan University, Nepal for the award of the degree of Doctor of Philosophy (Ph.D.), is a research work carried out by me under the supervision of Prof. Dr. Rejina Maskey Byanju, Central Department of Environmental Science, Tribhuvan University and co supervised by Dr. Prakash V. Bhave (Duke University, USA).

This research is original and has not been submitted earlier in part or full in this or any other form to any university or institute, here or elsewhere, for the award of any degree.

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ABSTRACT

Air pollution is a serious issue in Nepal. It has an impact on the environment as well as human health. This study covers three main sectors of air pollution: crop residue open burning (CROB), open burning of municipal solid waste (MSW), and emissions from diesel vehicles. To estimate specific pollutants from various sources, an inventory of emissions factors (EFs) is required.

This study presents a gridded emissions inventory of Nepal's key open burning sectors at a fine resolution of 1 km x 1 km. In 2016/17, the mass of CROB was 2,908 Gg, accounting for 22% of the dry matter produced. The air pollutant emissions were calculated by multiplying the mass of crop residue burned with EFs, which was CO_2 4,140, CO 154, CH_4 6.5, SO_2 1.2, $PM_{2.5}$ 24.5, OC 8.6, BC 2.2, NO_x 7, NMVOC 22.5, and NH_3 2.7, in Gg/yr. Open burning was more common in districts with less cattle per hectare. The other contributing factors were the use of combine harvesters, especially in the Tarai districts, and labor migration. Rearing more cattle and utilizing dung for clean energy production as well as industrial raw materials could be the mitigation options to reduce CROB. Likewise, the entire quantity of garbage burnt openly in 2011 was estimated to be 89.2 Gg (i.e., 0.24 Gg/day), accounting for 4.5% of all waste generation. By multiplying the mass of waste burnt with EFs, the air pollutant emissions were estimated as $PM_{2.5}$ 0.67 (OC 0.51), PM_{10} 0.72, BC 0.3, CO_2 145, CH_4 0.36, SO_2 0.06, NO_x 0.23, CO 7.66, NMVOC 1.36, and NH_3 0.07, all in Gg. Lower waste collection efficiency reveals more open burning, especially in the rural areas of Kathmandu Valley and Nepal. MSW open burning could be minimized with proper waste collection services as well as recycling practices.

The emission inventory of diesel vehicle categories was obtained through experimentally-based EFs for the Kathmandu Valley and Nepal. The fuel-based EFs of CO_2, CO, BC, and $PM_{2.5}$ were calculated using the carbon mass balance technique. In average, the EFs of the diesel vehicles measured while idling (n = 29) were 2,600 for CO_2, 33.3 for CO, 0.6 for BC, and 5.2 for $PM_{2.5}$, in unit of g/L. In average, the EFs of the diesel vehicles measured while moving (n = 5) were 2,476 for CO_2, 97.3 for CO, 1.7 for BC, and 20.7 for $PM_{2.5}$, in g/L. In 2017/18, CO_2 was estimated as 2,214-2,781, CO 27.7-88.8, BC 0.51-3.55 and $PM_{2.5}$ 3.42-23.47, in Gg, of which 24.4-29.5%, 28.9-32.3%, 12.3-31.9%, and 21.8-42.5% was respectively for the Kathmandu Valley. Higher emission were from lower euro-grade vehicles, including lack of proper maintenance in time, low fuel quality, traffic congestion, and roadway-grade. Implementation of higher grade vehicles (> Euro III/BS III), electric vehicles, better fuel quality with low sulfur, timely repair and maintenance of vehicles, road repair and opting for low-carbon sustainable pathways could significantly reduce air pollution from the transport sector. This study recommends sound policies for open burning and transport sectors that would significantly improve air quality as well as assist in minimizing negative health effects in Nepal.

Keywords: Open burning sectors; Diesel vehicles; Emission factors; Emission inventory; Policies

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LIST OF SYMBOLS

B_l : Quantity of agricultural residue that can be burned

BS : Bharat standard

C_A : Amount of crop residue used as animal fodder

C_F : Amount of crop residue used as energy

C_T : Amount of crop residue used to thatch the roof

D_1 : Ratio of dry matter to crop residue for a crop type l

E : Emission

F : Fuel consumption

Gg : Gigagram

g/L : Gram per liter

h : Hour

kL : Kilo liter

km : Kilometer

mm : Millimeter

M : Mass

M_P : Molecular weight of pollutant P

N_1 : Ratio of residue to crop for a crop type l

P_1 : Quantity of crop production for a crop type l

P : Pressure of a gas

[p] : Exhaust concentration of pollutant P

P_{frac} : Fraction of population burning MSW

R : Universal gas constant

s : Second

Tg : Teragram

T : Temperature of a gas

V : Volume of a gas

w_c : Fuel's carbon weight fraction

ρ_f : Density of fuel

η : Burn efficiency fraction/burning oxidation efficiency

δ : Waste combustible fraction

λ : Fraction of MSW burning at disposal sites

$\mu g/m^3$: Microgram per cubic meter

LIST OF ACRONYMS AND ABBREVIATIONS

ADB : Asian Development Bank

BC : Black Carbon

CAGR : Compound Annual Growth Rate

CBS : Central Bureau of Statistics

CDES : Central Department of Environmental Science

CH_4 : Methane

CO : Carbon Monoxide

CO_2 : Carbon Dioxide

CROB : Crop Residue Open Burning

CV : Coefficient of Variation

DM : Dry Matter

DoTM : Department of Transport Management

EF : Emission Factor

FID : Flame Ionization Detector

FIRMS : Fire Information for Resource Management System

GDP : Gross Domestic Product

GFED : Global Fire Emissions Database

GIS : Geographic Information System

GWPs : Global Warming Potentials

HEPA : High Efficiency Particulate Air

HC : Hydro Carbon

HD : Heavy Duty

H_2S : Hydrogen Sulfide

HDDV : Heavy Duty Diesel Vehicles

ICIMOD : International Centre for Integrated Mountain Development

IPCC : Intergovernmental Panel on Climate Change

LCV : Light Commercial Vehicles

LDDV : Light Duty Diesel Vehicles

MoAD : Ministry of Agriculture Development

MSW : Municipal Solid Waste
MSWGR : Municipal Solid Waste Generation Rate
NAAQS : National Ambient Air Quality Standard
NAST : Nepal Academy of Science and Technology
NDIR : Non Dispersive Infrared
NEEDS : Nepal Energy & Environment Development Services P. Ltd.
NOC : Nepal Oil Corporation
NVMES : Nepal Vehicle Mass Emission Standard
NH_3 : Ammonia
NMVOC : Non Methane Volatile Organic Carbon
N_2O : Nitrous Oxides
NO_x : Nitrogen Oxides
NO_2 : Nitrogen Dioxides
NO : Nitric Oxide
OC : Organic Carbon
O_2 : Oxygen
PM : Particulate Matter
PTFE : Poly Tetra Fluoro Ethylene
rpm : Revolutions per Minute
sccm : Standard Cubic Centimetres
SD : Standard Deviation
SO_2 : Sulphur Dioxide
SWM : Solid Waste Management
SWMTSC : Solid Waste Management and Technical Support Centre
THC : Total Hydro Carbon
TSP : Total Suspended Particles
VFTC : Vehicle Fitness Testing Centre
VIIRS : Visible Infrared Imaging Radiometer Suite
VKT : Vehicle Kilometer Travelled
VOC : Volatile Organic Carbon
WECS : Water and Energy Commission Secretariat

CHAPTER 1

INTRODUCTION

1.1 Background

Nepal is an Asian country that lies on the Himalayan mountain ranges' southern slopes. It is a landlocked country bordered by India on the east, west, and south, and the Tibet Autonomous Region of China on the north. The rural areas/districts in this study, refer to the difference between areas of 75 districts and 58 municipalities before the new constitution of Nepal. Nepal's 2015 Constitution establishes the country as a Federal Democratic Republic with seven provinces. There are 460 rural municipalities, 276 municipalities, 11 sub-metropolitan cities, and 6 metropolitan cities among the 753 local levels. The country is divided into 77 districts for administrative purposes (CBS, 2019). Until the restoration of zones to provinces, the Bagmati Zone (a deprecated geopolitical region that included the entire Kathmandu Valley) was one of Nepal's fourteen zones. Its headquarters was Kathmandu Valley. The bowl-shaped Kathmandu Valley lies at an elevation of 1,400 meters in the Himalayan foothills, surrounded by mountains and forests. The Kathmandu Valley's total urban area is 96.68 km^2 (KVDA, 2017), and it has Nepal's highest population density. Kathmandu Metropolitan City (KMC), Lalitpur Sub-Metropolitan City (LSMC), Bhaktapur, Kirtipur, and Madhyapur Thimi were the five erstwhile municipalities in the valley during the time of the study. Later, Nepalese government designated 16 new municipalities in the valley, partially in reaction to the rapidly growing urban population (KVDA, 2017). The Kathmandu Valley contains three types of roads: the inside ring road (i.e., urban city areas); the ring road (which encircle the entire valley and is connected to the outer ring road); and the outside ring road (which is connected to Nepal's highways).

Air pollution is an emerging issue in Nepal. According to the Health Effects Institute (2020), Nepal was ranked as the world's second most polluted country in terms of $PM_{2.5}$ concentration (i.e., an annual average of 83.1 g/m3) for the year 2019. Recently (from

March 26 to April 9, 2021), the Kathmandu Valley had the highest air pollution on record (i.e., globally first rank), where $PM_{2.5}$ incredibly exceeded the World Health Organization (WHO) guideline (i.e., a daily average above 200 $\mu g/m^3$) (AirNow Department of State, 2021). Moreover, other parts of Nepal also experienced severe air pollution. The government of Nepal had to declare a red alert/emergency condition for the entire nation. To realize the public health issues, schools were shut down from March 30th to April 3rd, 2021. Also, due to poor visibility, domestic and international flights were severely disrupted (The Himalayan, 2021).

Open burning of solid waste and biomass, and combustion technologies are the major contributing sources to the rise in air pollution (Sadavarte *et al.*, 2019; Shrestha, 2018). Those sources emit not only known air pollutants (CO, NOx, SO_2, NH_3, NMVOC), but also particulate matter (PM_{10}, $PM_{2.5}$, BC, OC) as well as greenhouse gases (CO_2, CH_4, N_2O) (Andreae and Merlet, 2001; Chang and Song, 2010; Dennis *et al.*, 2002; Gadde *et al.*, 2009; Hossain and Park, 2012; Sadavarte *et al., 2019*; Sahai *et al.*, 2007; Shrestha, 2018; Tipayarom and Oanh, 2007; Viana *et al.*, 2008; Zhang *et al.*, 2000).

Following their release into the atmosphere, air pollutants undergo additional chemical and physical transformations that might disrupt the earth's radiation balance and have a negative impact on air quality (Jethva *et al.*, 2019; Permadi and Oanh, 2013) and human health (Tipayarom and Oanh, 2007; Zhang *et al.*, 2017). Life-threatening particulate matter (PM) enriched in organic carbon (OC), and elemental carbon (EC), carcinogenic dioxins, and numerous other harmful pollutants like carbon monoxide (CO), nitrogen oxides (NO_x), sulfur dioxide (SO_2), and non-methane volatile organic compounds (NMVOCs) are all present in the emitted smoke (Guttikunda, 2007; Guttikunda *et al.*, 2014; Hodzic *et al.*, 2012; Nagpure *et al.*, 2015; Pokhrel and Viraraghavan, 2005; Wiedinmyer *et al.*, 2014). High levels of PM can cause respiratory and cardiovascular problems, as well as cancer and adverse birth outcomes (McDonnell *et al.*, 2000). In 2019, 42,100 people died in Nepal as a result of air pollution (HEI, 2020). Because the carbon dioxide (CO_2) and black carbon (BC) released by burning both have a significant global warming potential, it also exacerbates local and regional warming. Methane (CH_4)

is another strong greenhouse gas that is released as organic waste degrades. Open burning also emits ammonia (NH_3), which can contribute to the formation of PM in the atmosphere.

Despite numerous research studies measuring pollutant concentrations in Nepal, attributing ambient pollution to sources, analyzing meteorological patterns, developing emission inventories, assessing human exposure and health, and recommending government policies, our air quality management remains primitive (Sadavarte *et al.*, 2019; Shrestha, 2018). To account for the annual estimate of specific pollutants from various sources, an emission inventory was required (Pradhan *et al.*, 2012; Sadavarte *et al.*, 2019). The sectoral emissions were estimated by multiplying activity data with country-specific, regional, and global emission factors (EFs). The EF is usually expressed as the amount of pollutants released per unit of fuel consumed (i.e., g/kg or g/L) (Defra, 2009; IPCC, 2006; Jayarathne *et al.*, 2018; Shrestha *et al.*, 2013a; Shrestha, 2018; Stockwell *et al.*, 2016).

Air pollution emissions from agricultural residue burning were estimated from 2003/04 to 2016/17 using secondary sources of information on crop production, residue consumption patterns, crop residue burning parameters, and EFs. Using the Geographic Information System (GIS) and fire counts as a proxy for temporal estimation, a gridded model-ready emission inventory was developed. A computational tool was used to develop a model-ready emissions inventory (1 km x 1 km) for agriculture residue open burning in Nepal. Likewise, emissions from MSW open burning for Kathmandu valley municipalities were estimated through a household survey on waste management, a transect walk survey, a waste combustible fraction experiment, a survey on the fraction of people burning trash outside households, and a survey on MSW burnt fraction at dumping sites. Other parameters were obtained from the published literature, including burning and oxidation efficiency, municipal populations, MSW generation rates, and EFs. These parameters were further utilized to estimate national emissions from 2001-2016. A complete gridded model-ready emissions inventory (1 km x 1 km) for MSW open burning was developed in Nepal.

Furthermore, using national statistics data, average vehicle kilometers traveled, fuel mileage, and experimental-based EFs for each vehicle category during idling and moving circumstances, this study calculated emissions from diesel vehicles from 1989/90 to 2017/18. Instruments like Ratnoze1 and Microaeth AE51 were used to measure emissions. The carbon mass balance method was used to calculate fuel-based EFs for CO_2, CO, BC, and $PM_{2.5}$. Lastly, the available policies, acts, guidelines, standards, and practices were thoroughly reviewed and gaps were identified. Policy suggestions were made based on the findings of this study as well as the global context.

1.2 Rationale

Air pollution is a major source of worry, owing to poor air quality, human exposure, public health effects, reduced visibility, and climate change. Despite being a serious issue, there are only limited studies so far in the world that consider a gridded model-ready emission inventory of open burning sectors (Chang and Song, 2010; Chang *et al.*, 2013; Pandey *et al.*, 2014; Venkataraman *et al.*, 2006). A high-resolution (1 km x 1 km) study for Nepal was just completed (Sadavarte et al., 2019); however, it did not include the open burning sectors. The Global Fire Emissions Database provided a gridded inventory for crop residue open burning; however, it was of coarse resolution (0.25° x 0.25°) and not suitable for a simulation in Nepal. In addition, there are information gaps and significant uncertainties in the emissions estimation from this sector, as detailed in section 2.1.1 (Chang *et al.*, 2013; Gadde *et al.*, 2009; Mona and Hitesh, 2016). The Global Fire Emissions Database's gridded inventory for open burning of agriculture residue currently has a coarse resolution (0.25° x 0.25°), which was insufficient for a simulation in Nepal. This study tried to construct a model-ready emission inventory (1 km x 1 km) for open burning sectors in order to better understand the geographical and temporal aspects of air pollutant emissions.

Research on on-road vehicle exhaust emissions was undertaken in Kathmandu more than two decades ago, with the study focused only on CO and HC emissions (Zhang *et al.*, 1995). Shrestha *et al.* (2013b) investigated the vehicle fleet in the Kathmandu Valley in order to estimate the environmental and climate co-benefits of technological advancements. The model employed, however, was the International Vehicle Emissions Model (IVE). Sadavarte *et al.* (2019) also released an emission inventory for the transportation sector; however, the analysis relied entirely on secondary data, and the EFs were calculated using global-based values. No studies have taken into account country-specific EFs, resulting in considerable uncertainty in Nepal's transportation sector emission inventory. This study focused on fuel-based EFs for idle and on-road diesel vehicles for Nepal. An emission inventory for diesel vehicles was developed for the Kathmandu Valley and Nepal using experimental-based EFs. Based on the existing literature, this research addresses worldwide and national policies and practices for vehicular emissions control. The findings of this study, as well as lessons learned from other nations, will aid policymakers in developing effective policy packages for air pollution control in the Kathmandu Valley and other regions of Nepal.

1.3 Objectives

1.3.1 General objective

The overall objective of this study was to develop emission inventory of Nepal’s air pollutants for effective air quality management.

1.3.2 The specific objectives

I. Development of a gridded, model-ready emission inventory of all major air pollutants from the major open burning sectors in Nepal.
II. Development of a fuel-based emission inventory of air pollutants from road vehicles in the Kathmandu valley.
III. Establishment of sound scientific information to review the policy for reducing vehicular air pollution in the Kathmandu Valley.

CHAPTER 2

DEVELOPMENT OF A GRIDDED, MODEL-READY EMISSION INVENTORY OF THE OPEN BURNING SECTORS IN NEPAL

Abstract

Open burning emissions are poorly characterized and under-estimated sources of air pollution in developing countries, including Nepal. This study includes open-burning sectors such as agriculture and municipal solid waste (MSW), which are major outdoor contributing sources of air pollution in Nepal. The Global Fire Emissions Database's gridded inventory for open burning of agriculture residue currently has a coarse resolution (0.25° x 0.25°), which was insufficient for a simulation in the country. This study developed a fine-resolution (1 km x 1 km), gridded model-ready emissions inventory for agricultural residue open burning to better understand the geographical characteristics of air pollutant emissions. From 2003 to 2017, air pollution emissions from agricultural residue burning were quantified at a national level. Secondary sources provided information on crop production, residue consumption patterns, crop residue burning parameters, and EFs. The amount of agricultural residue burned in 2016/17 was 2,908 Gg (61-139%), accounting for 22% of the dry matter generation. The emissions were calculated by multiplying the mass of crop residue burned with EFs, which was CO_2 4,140 (56-144%), CO 154 (4-196%), CH_4 6.5 (7-193%), SO_2 1.2 (60-140%), $PM_{2.5}$ 24.5 (30-170%), OC 8.6 (38-162%), BC 2.2 (-1-201%), NO_x 7 (54-146%), NMVOC 22.5 (8-192%) and NH_3 2.7 (3-197%), in Gg in 2016/17. From February to May, over 80% of emissions were produced, with April being the peak month.

Similarly, this work established a complete gridded model-ready emissions inventory (1 km x 1 km) for MSW open burning in Nepal for the first time. Nationally, air pollution emissions were estimated from 2001-2016. The population surrogate/spatial data (1 km x 1 km) was available for the year 2011. Also, many municipalities' data were collected around the same year. Therefore, the year 2011 was considered a base year in this study.

The study in the Kathmandu Valley included a survey of household, a survey through transect walk, a waste combustible fraction experiment, a survey on the fraction of the people burning trash outside households, and a survey on MSW burnt fraction at dumping sites. Other parameters were taken from the literature, including burning/oxidation efficiency, municipal populations, MSW generation rates, and EFs. These parameters were also utilized to estimate national emissions. After multiplying the burned MSW mass by EFs, the emissions for five erstwhile municipalities of the Kathmandu Valley were estimated as $PM_{2.5}$ 0.06 Gg (OC 0.04 Gg and EC 0.001Gg), PM_{10} 0.06 Gg, BC 0.03 Gg, CO_2 11.9 Gg, CH_4 0.03 Gg, SO_2 0.005 Gg, NO_x 0.02 Gg, CO 0.63 Gg, NMVOC 0.11 Gg, and NH_3 0.006 Gg in 2016. In 2011, the total amount of trash burnt in the Nepal was 89 Gg (i.e., 0.24 Gg/day), accounting for around 4.5% of all waste generated that year. Over the last decade, there has been a 46% rise in MSW open burning. In 2011, 8.8 Gg of MSW were burned in urban areas (58 municipalities), whereas 80 Gg were burned in rural regions (75 rural districts). According to this study, the yearly trend of open burning in urban areas was much greater (350%) than in rural regions from 2001 to 2016.

According to the conclusions of this research, a considerable reduction in open burning might enhance air quality in Nepal and assist minimize health effects like cardiovascular disease, acute as well as chronic respiratory disorders, allergic hypersensitivity, and premature mortality, as well as climate impacts.

Keywords: Crop residue open burning; MSW open burning; Emission factors; Gridded emission inventory; Health impacts

2.1 Crop Residue Open Burning

2.1.1 Introduction

Open burning of agricultural waste is seen as a severe problem that contributes significantly to air pollution in Asia and across the world (Streets *et al.*, 2003; Tipayarom & Oanh, 2007). Crop residue open burning (CROB) is a common occurrence in many nations, particularly during harvesting or dry seasons (Chang *et al.*, 2013). After coal, oil, and natural gas, it is the fourth most common fuel globally, and it is utilized by roughly half of the world's population, primarily in developing nations (Guoliang *et al.*, 2008). CROB and its influence on the atmosphere and climate have been linked in several research (Venkataraman *et al.*, 2006). Trace Gases (CO, NO_x, NMVOC, $SO_{2,}$ NH_3), greenhouse gases (N_2O, CO_2, CH_4), and particulate matter ($PM_{2.5,}$ PM_{10}, BC, OC) are released when agricultural residue is burned openly (Andreae and Merlet, 2001; Chang and Song, 2010; Dennis *et al.*, 2002; Gadde *et al.*, 2009; Hossain and Park, 2012; Sahai *et al.*, 2007; Shrestha, 2018; Tipayarom and Oanh, 2007; Viana *et al.*, 2008; Zhang *et al.*, 2000).

The influence of greenhouse gases on air quality is detrimental and have the ability to change the earth's radiation balance (Jethva *et al.*, 2019; Permadi and Oanh, 2013) as well as public health (Tipayarom and Oanh, 2007; Zhang *et al.*, 2017). CROB has been shown to have an influence on weather and atmospheric visibility (Chauhan and Singh, 2017), unfavorable health outcomes such as asthma attacks in children (Torigoe *et al.*, 2000), health-related economic impact (Li *et al.*, 2017), and decline of a country's economy.

Several studies on CROB emission inventories have been carried out over the world (Andreae and Merlet, 2001; Gadde *et al.*, 2009; Irfan *et al.*, 2014; Li *et al.*, 2017; Ni *et al.*, 2015; Sahai *et al.*, 2011; Shrestha, 2018; Streets *et al.*, 2003; Yang *et al.*, 2008; Zhang *et al.*, 2008); however, research on gridded model-ready emission inventories is scarce (Chang and Song, 2010; Chang *et al.*, 2013; Pandey *et al.*, 2014; Venkataraman *et al.*, 2006). This is owing to high uncertainty and information gaps in emissions estimation from open burning of agricultural residue (Chang *et al.*, 2013; Gadde *et al.*, 2009; Mona and Hitesh, 2016). Furthermore, the percentage of agricultural residue burned in fields is not well known globally, which causes further uncertainties (Venkataraman *et al.*, 2006).

Using computational tools such as Geographic Information System (GIS), this study attempted to develop a gridded model ready emission inventory (1 km x 1 km) (Markakis *et al.*, 2013). Even though there are alternative ways for determining global biomass burning, such as remote sensing, it has not been frequently employed in Asian nations due to several cautions and limitations (Chang *et al.*, 2013). First, if the area of the cropland and active fire sites are lower than satellites' spatial scale (i.e., a resolution of less than 1 km^2), mixed pixel difficulties in picture categorization may occur (Chang *et al.*, 2013). Second, short-lived crop fires are not detectable by satellite repetition cycles. Active fire data is not taken into consideration in this situation, resulting in a higher level of uncertainties in the outcome (Kanabkaew and Oanh, 2011; Vander-Werf *et al.*, 2010). According to Benali *et al.* (2016) orbiting satellites do not identify active flames of short duration (< 12 h). Third, the presence of prolonged cloud cover and many fire incidents inside a single fire perimeter add to the uncertainty. Finally, despite the fact that the Global Fire Emissions Database (GFED) is an extensively used, it does not account for many minor fires (Zhang *et al.*, 2018), resulting in uncertainty. Presently, only coarse-resolution (0.25° x 0.25°) model-ready emission inventory (Bhardwaj *et al.*, 2018; Kurokawa *et al.*, 2013; Rupakheti *et al.*, 2017) and at fine-resolution (1 km x 1 km) (Sadavarte *et al.*, 2019) exists; nevertheless, CROB is excluded. This study attempted to construct a robust gridded emission inventory (1 km x 1 km) from open burning of crop residue for the year 2016/17 and projected between 2003/04 and 2016/17 for Nepal.

2.1.2 Objectives

The general objective of this research was to develop a CROB emission inventory for Nepal.

The specific objectives were:

I. To calculate crop residue open burning parameters for CROB estimation.
II. To estimate amount of residue burned openly in the agricultural field in Nepal.
III. To compute EFs based on secondary sources and estimate air pollutants from CROB in Nepal.
IV. To spatially allocate emission flux from CROB at 1 km x 1 km fine resolution.

2.1.3 Materials and methods

2.1.3.1 Crop production

Rice, wheat, millet, maize, oil crops, barley, potato, jute, sugarcane, and pulses (i.e., lentil, pigeon pea, chick pea, grass pea, black gram, horse pea, and soybean) were among the crop varieties studied. The district-wise data of this type from 2003 - 2017 (tons/yr) was collected from annual reports of MoAD (2003 - 2017), and from it crop production was estimated for Nepal.

2.1.3.2 Crop residue generation

The residue to crop ratio was used from various sources to calculate agricultural residue: for rice, Jain *et al.* (2014), Sahai *et al.* (2011), and Singh and Gu (2010) were used; likewise maize from Koopmans and Koppejan (1997), Singh and Gu (2010), and Yang *et al.* (2008); wheat and millet from Sahai *et al.* (2011) and Singh and Gu (2010); oil crops from Yang *et al.* (2008); barley from Singh and Gu (2010); potato from Yang *et al.* (2008); sugarcane from Global Atmospheric Pollution Forum (2010), and Singh and Gu (2010); jute from Koopmans and Koppejan (1997); and pulses from Yang *et al.* (2008) were used (Table 2.1).

Dry matter from residue generation was further calculated from the dry matter to crop residue ratio which was taken from the literature: rice from Streets *et al.* (2003); maize, barley, wheat, and potato from IPCC (1996); millet from GAPF (2010); oil crops from Jain *et al.* (2014), and Bhattacharya and Mitra (1998); pulse from Shrestha (2018); and jute and sugarcane from Bhattacharya and Mitra (1998) (Table 2.1).

Total dry matter was estimated by multiplying the quantity of crop production by the residue to crop ratio and the dry matter to crop residue ratio (Eq. 2.1), which was obtained from IPCC (1996), Kanabkaew and Oanh (2011), Shrestha *et al.* (2013a), and Shrestha (2018).

$$C_R = P_1 \times N_1 \times D_1 \quad (2.1)$$

Where, C_R represents total dry matter (in kg); P_1 represents amount of crop production (in kg) (for crop type l); N_1 represents residue to crop ratio (for crop type l); and D_1 represents the dry matter to crop residue ratio (for crop type l).

2.1.3.3 Quantity of crop residue burnt in the agriculture field

The quantity of agriculture residue accessible for open burning (Eq. 2.2) following its consumption pattern (i.e., animal food, roof thatching, and fuel energy) was acquired from Sahai *et al.* (2011). For the regions where there was a residual utilization shortfall, we assumed that open burning was negligible.

$$B_l = C_R - (C_F + C_T + C_A) \quad (2.2)$$

Where, B_l represents amount of agriculture residue (in kg) accessible for open burning; C_F represents agriculture residue (in kg) utilized in the form of fuel energy; C_T represents agriculture residue (in kg) used for roof thatching; and C_A represents agriculture residue (in kg) utilized in the form of animal fodder.

Data on fuel energy of Nepal (i.e., Gigajoule) was gathered between 2003/04 and 2008/09 (WECS, 2010) and 2014/15 (WECS, 2017) and converted to metric tons using WECS (2014). The fuel energy value was estimated using linear regression for rest of the years (2009/10-2013/14; 2015/16-2016/17). The weightage of residue generation was used to distribute these national values at the district level. In Nepal, no data on the proportion of agricultural residue utilized for roof thatching has been reported yet (Das *et al.*, 2020). Its value was extrapolated from India, assuming that the two countries' values would be comparable. According to the available literature, 2% of total rice residue was utilized for house thatching in India (Venkataraman *et al.*, 2006). To estimate the amount of residue used as animal feed, data on animal census and fodder needs were acquired from the available literature.

The statistics on district-level animal censuses from 2003/04 to 2016/17 was obtained from the Annual Reports of MoAD (2003/04-2016/17). We looked at main local animal kinds such as total buffalo, total cattle, milking buffalo, and milking cows in our study. The district-level animal census from 2003/04 to 2016/17 was taken from Annual Reports of MoAD (2003/04-2016/17).

Buffalo and non-milking cattle (i.e., young stock, adult males and dry) were expected to be similar to that of India (Dikshit & Birthal, 2010). This method was used to estimate Nepal's comprehensive buffalo and cow census. Dikshit and Birthal (2010) provided the dry matter requirements for buffalo and cattle (i.e., young stock, adult males, dry, and milking) on a daily basis (Table 2.2). The overall dry matter demand was estimated based on it from 2003/04 - 2016/17. Furthermore, the periodic feed consumption was taken into account from animal's feed inventory of Nepal (NAFLQML, 2019).

The amount of residue burned (M_1) in the cropland was calculated by multiplying B_l by η (burn efficiency fraction) (IPCC, 1996; Kanabkaew & Oanh, 2011; Shrestha *et al.*, 2013a; Shrestha, 2018).

$$M_l = B_l \times \eta \quad (2.3)$$

Referring to Turn *et al.* (1997), the values of η (in fraction) for maize, rice, wheat, sugarcane, and barley were gathered. Likewise from IPCC (1996), the values of η for oil crops, millet, tobacco, potato, and jute were obtained. For pulses, Shrestha (2018) was considered for η. The average value of η was determined (Table 2.1) and applied in eq. (2.3) to estimate the amount of residue burned openly.

Table 2.1: Crop residue open burning parameters obtained from various literatures.

	Rice	Maize	Millet	Wheat	Barley	Oil crops	Potatoes	Sugarcane	Jute	Pulses	Average
Residue to crop ratio (N)	1.5[a,b,c]	2[a,d,e]	1.2[a,b]	1.5[a,b]	1.3[a]	2[e]	0.5[e]	0.33[a,h]	2[i]	1.5[e]	-
Dry matter to crop residue ratio (D)	0.85[j]	0.4[k]	0.8[h]	0.83[k]	0.83[k]	0.8[c,l]	0.45[k]	0.8[l]	0.8[l]	0.8[g]	-
Burn efficiency fraction (η)	0.89[m]	0.92[m]	0.9[k]	0.86[m]	0.82[m]	0.9[k]	0.9[k]	0.68[m]	0.9[k]	0.9[g]	0.86±0.07[n]

[a] Singh and Gu, 2010
[b] Sahai *et al.*, 2011
[c] Jain *et al.*, 2014
[d] Koopmans and Koppejan, 1997
[e] Yang *et al.*, 2008
[g] Shrestha, 2018
[h] GAPF, 2010
[i] Koopmans and Koppejan, 1997
[j] Streets *et al.*, 2003
[k] IPCC, 1996
[l] Bhattacharya and Mitra, 1998
[m] Turn *et al.*, 1997
[n] calculation

Table 2.2: Dry fodder requirement (kg/animal/day).

	In milk	Dry	Adult male	Young stock
Cattles	5.5	4.02	6.03	2.13
Buffalo	6.34	4.95	7.47	2.22

Source: Dikshit & Birthal, 2010

2.1.3.4 Emission factors and emissions estimation

This study employed the available literatures of country-based EFs (Jayarathne *et al.*, 2018; Stockwell *et al.*, 2016). In some circumstances EFs from other nations were used, especially when its value were unavailable (Andreae and Merlet, 2001; Cao *et al.*, 2008; Dennis *et al.*, 2002; Kanabkaew and Oanh, 2011; Streets *et al.*, 2001; Turn *et al.*, 1997). When EFs were reported with multiple values in most datasets, they were averaged (Table 2.3). Table 2.4 contains a list of the sources from which it was gathered. The average EFs as well as standard deviation (SD) was estimated for individual pollutant referring crop types (Fig. 2.1 & Table 2.3). These values were used to calculate the emission trends. For the 2016-17 fiscal year, we considered the most comprehensive district-wise emission inventory.

Emissions 'E in kg' for pollutant type 'i' and crop type 'l' were calculated referring eq. (2.4), in which 'M_1 in kg' from eq. (2.3) was multiplied by '$EF_{i,l}$ g/kg' of pollutant type 'i' from crop type 'l' (IPCC, 2006; Kanabkaew & Oanh, 2011; Shrestha *et al.*, 2013a; Shrestha, 2018).

$$E_{I,l} = \sum_{l} M_l \times EF_{I,l} \quad (2.4)$$

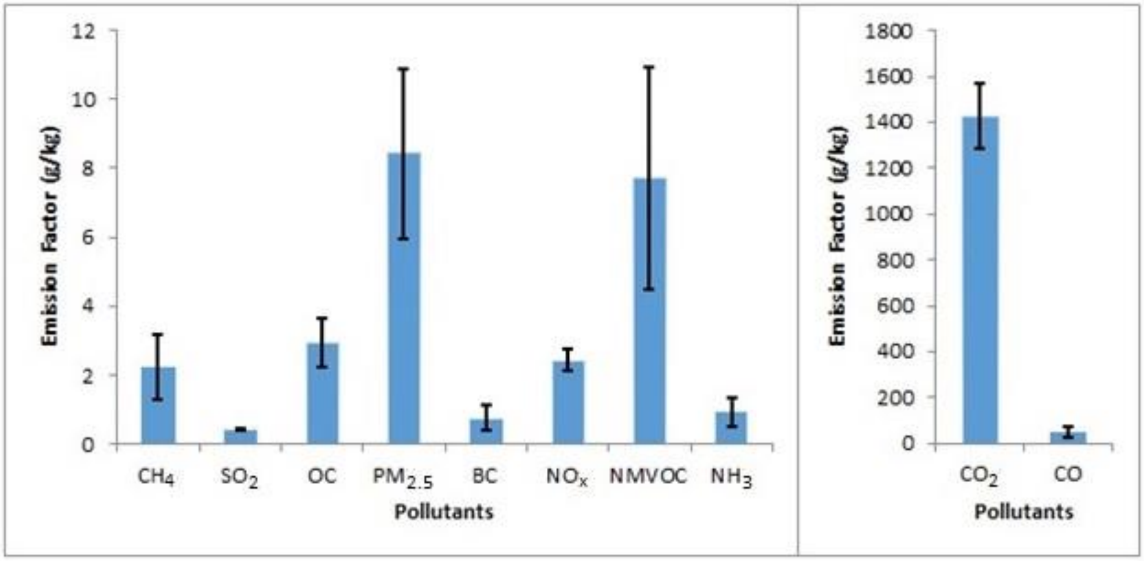

Figure 2.1: Selected best value of EFs with standard deviation (compilation from country-specific, regional and global values)

Table 2.3: EFs (g/kg) for estimation of emissions from open burning of crop residue

	Rice	Maize	Millet	Wheat	Barley	Oil crops	Potatoes	Sugarcane	Jute	Pulse	Avg. EFs (SD)
CO_2	1474[q]	1322±45[i]	1613[s]	1445[q]	1445[y]	1395[q]	1613[s]	1130[zb]	1345[ze]	1470[zl]	1425±(142)
CO	31.4[q]	60.4±28.9[h]	36.4[u]	49.4[q]	49.4[y]	77.0[q]	36.4[c]	34.7[u]	106[ze]	47.3[zl]	52.8±(23.2)
CH_4	1.2[q]	3.0±2[j]	2.7[s]	1.9[q]	1.9[y]	3.0[q]	2.7[s]	0.4[u]	3.6±1.3[zf]	1.9[zl]	2.2±(1.0)
SO_2	0.4[zj]	0.4[s]	0.4[s]	0.5[zk]	0.5[y]	0.4[s]	0.4s	0.4[s]	0.4[s]	0.4[y]	0.4±(0.02)
OC	4.2±3[c]	3.3±1.8[m]	2.9±0.5[v]	2.7±1.5[f]	2.7±1.5[y]	2.5±0.8[z]	2.9±0.5[v]	1.5[zc]	3.6±2.2[zg]	3.3[s]	3.0±(0.7)
$PM_{2.5}$	11.1±3.2[b]	8.5±4[n]	7.7±3.8[w]	6.7±3.0[e]	6.7±3.0[y]	11.5[za]	7.7±3.8[w]	3.8±0.1[zd]	9.2±4.7[zh]	11.5[zm]	8.4±(2.5)
BC	0.3[q]	0.7[o]	0.7[s]	0.6[q]	0.6[y]	1.7[q]	0.7[s]	0.6±0.1[zc]	0.7[s]	0.8[zl]	0.8±(0.3)
NO_x	2.8±0.9[a]	2.4±1.6[p]	2.5±0.6[x]	1.9±0.9[d]	1.9±0.9[y]	2.5±0.6[x]	2.5±0.6[x]	2.6[u]	2.6±0.6[zi]	2.5[s]	2.4±(0.3)
NMVOC	7.0[s]	7.2±4[r]	11.1±4.9[k]	4.8±3.8[g]	4.8±3.8[y]	11.1±4.9[k]	11.1±4.9[k]	2.2[u]	11.1±4.9[k]	7.0[s]	7.7±(3.2)
NH_3	0.2[q]	0.9±0.3[t]	1.3[s]	0.7[q]	0.7[y]	1.5[q]	1.3[s]	1.0[u]	1.3[s]	0.6[zl]	0.9±(0.4)

The values below are obtained from Table 2.4.

a=avg. from ab, ac and ad; b= avg. from ae, af, ag and ah; c= avg. from ae, af, ag, ah, zc and ze; d= avg. from u, ab, ab, ae, ai and ak; e=avg. from u, ae, ag, ah and ai; f= avg. from o, ae, ag, ah, ai, ak, zc and ze; g= avg. from u and ai; h= avg. from u, ab, ag, ai, an and ze; i= avg. from ab, ag, ai and an; j= avg. from u and ai; k= avg. from s and ai; m= avg. from ae, ag, ai, zc and ze; n= avg. from u, ae, ag and ai; p= avg. from ab, ai and u; r= avg. from u and ai; t= avg. from s, u and ai; v= avg. from s, ap and zc; w= avg. from s, ah, ap and zc; x= avg. from s, ak, ap, aq and ze; z= avg. from s, ae, ap and zc; za= avg. from s, ae, ah, ap, zc and zm; zd= avg. from u and zc; zf= avg. from s and zb; zg= avg. from s, ae, ap, zc and ze; zh= avg. from s, ae, ap, zb and zc; zi= avg. from s, ak, ap, aq and ze; zj= avg. from ac, an and ze; zk= avg. from ai, an and ze; zl= avg. value of q; zm= avg. value

Table 2.4: Compiled EFs for crop straw/residues.

[o] Streets *et al.*, 2001	BC (maize)	China (combined value from the literature)
[q] Stockwell *et al.*, 2016	CO_2, CO, CH_4, BC, NH_3 (rice, wheat, oil crops); average EFs of CO_2, CO, CH_4, BC, NH_3 (pulses)	Nepal (field experiment based value from NAMaSTE campaign)
[s] Andreae and Merlet, 2001	NMVOC (rice); SO_2 and NH_3 (maize); CO_2, CH_4, BC, NH_3, SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (millet and potato); SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (oil crops); SO_2 (sugarcane); SO_2, BC, NH_3, NMVOC (jute); OC, NO_x, NMVOC (pulses)	Global-based compiled value
[u] Dennis *et al.*, 2002	CO, CH_4, $PM_{2.5}$, NO_x, NMVOC, NH_3 (maize); CO (millet and potato); $PM_{2.5}$, NO_x, NMVOC (wheat); CO, CH_4, NO_x, NMVOC, NH_3, $PM_{2.5}$ (sugarcane); OC, $PM_{2.5}$ (jute)	United States
[y] best assumption	-	-
[ab] Zhang et al., 2008	NO_x (rice); CO_2, CO, NO_x (maize); NO_x (wheat); $PM_{2.5}$, NO_x (jute)	China (laboratory based value)
[ac] Irfan *et al.*, 2014	SO_2, NO_x (rice)	Pakistan (field experiment based value)
[ad] Guoliang *et al.*, 2008	NO_x (rice)	Rural China (laboratory based value using a combustion chamber)
[ae] Li *et al.*, 2017	OC, $PM_{2.5}$ (rice, maize, wheat, oil crops); CH_4 (jute)	China (laboratory based value)
[af] Oanh *et al.*, 2011	OC, $PM_{2.5}$ (rice)	Thailand (in-situ experiment and laboratory hood experiment)
[ag] Ni *et al.*, 2015	OC, $PM_{2.5}$ (rice, wheat); CO_2, CO, OC, $PM_{2.5}$ (maize); CH_4, $PM_{2.5}$, NO_x (jute).	China (laboratory based value using a combustion chamber)
[ah] Hays *et al.*, 2005	OC, $PM_{2.5}$ (rice, wheat); $PM_{2.5}$ (millet, potato, oil crops); CH_4 (jute)	United States (experiment based values)
[ai] Li *et al.*, 2007	CO_2,CO, CH_4, SO_2, OC, $PM_{2.5}$, NO_X, NMVOC, NH_3 (maize); NMVOC (millet, potato); SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (wheat); NMVOC (oil crops); CH_4, OC, $PM_{2.5}$, NO_x, NMVOC (jute)	Rural China (experimental value using middle volume samplers)
[ak] Sahai *et al.*, 2007	NO_x (millet, potato, oil crops); OC, NO_x (wheat); CH_4 (jute)	India (in-situ experiment of wheat straw burning in a combine harvested crop field)
[al] Zhang *et al.*, 2000	NO_x (wheat)	Rural China (experiment based on residue fuel and stove combination)
[an] Jenkins *et al.*, 1996	SO_2 (rice, wheat); CO_2, CO (maize); $PM_{2.5}$, NO_x (jute)	United States (wind tunnel simulation)
[ap] Akagi *et al.*, 2011	OC, $PM_{2.5}$, NO_x (millet, potato, oil crops)	Global-based (laboratory based value through flaming and smoldering)
[aq] de Zarate *et al.*, 2000	NO_x (millet, potato, oil crops)	Spain (combustion chamber experiment)
[zb] Kanabkaew and Oanh, 2011	CO_2 (sugarcane)	Thailand
[zc] Turn *et al.*, 1997	OC (rice, maize, wheat); OC, $PM_{2.5}$ (millet, potato, oil crops); OC, BC, $PM_{2.5}$ (sugarcane); CH_4 (jute)	United States (laboratory based values)
[ze] Cao *et al.*, 2008	SO_2, OC (rice, wheat); CO, OC (maize); NO_x (millet, potato, oil crops); CO_2, CO, CH_4, $PM_{2.5}$ (jute)	China (laboratory based values)
[zm] Jayarathne *et al.*, 2018	$PM_{2.5}$ (oil crops, pulses)	Nepal (experiment, NAMaSTE campaign)

2.1.3.5 Spatial and temporal distribution

More complex computational techniques like GIS as well as fire counts as a proxy for temporal estimation were required to develop a gridded model-ready emission inventory (Markakis *et al.*, 2013; Ni *et al.*, 2015). For the year 2016/17, a software-based model was used to create a high-resolution emissions inventory (1 km x 1 km) for agricultural residue open burning in Nepal. The following datasets were required: outline boundary of Nepal; district boundary of Nepal; land-cover data of 2010 from ICIMOD (2013) (Fig. 2.2); and crop production statistics of 2016/17 in district-wise. The emissions from CROB were calculated referring the percentage of agricultural land under each grid of district. To better understand the BC emission flux, geographical distributions were constructed across the country. Similarly, fire counts obtained from the Visible Infrared Imaging Radiometer Suite (VIIRS) on cropland (FIRMS, 2020) were used to establish the temporal distribution of emissions (Ni *et al.*, 2015).

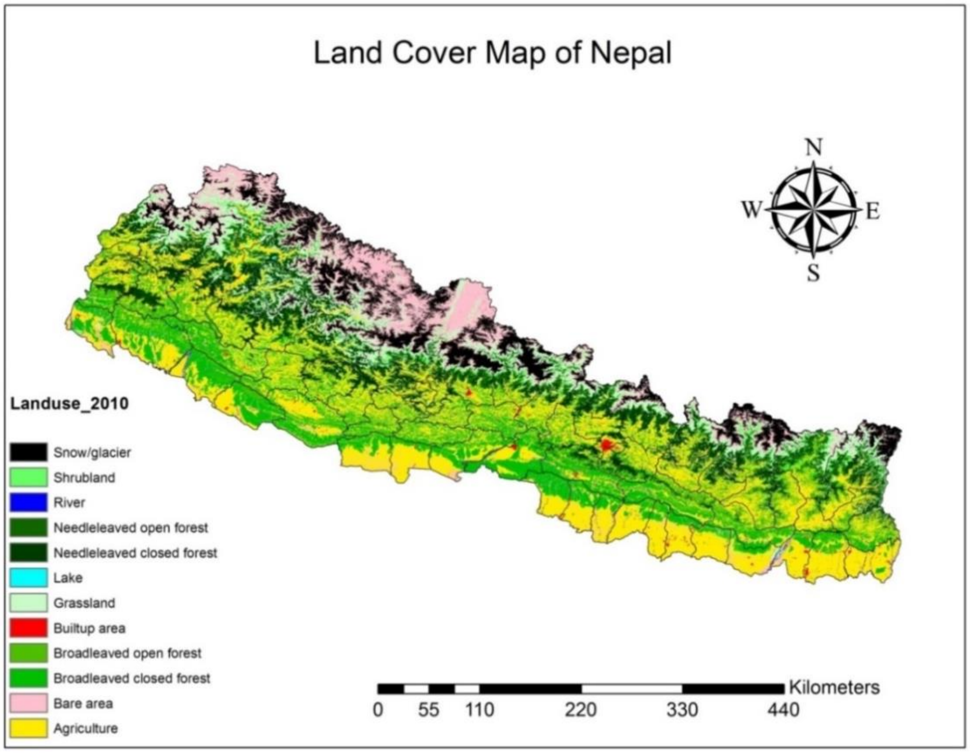

Figure 2.2: Land cover map of Nepal. (Source: ICIMOD, 2013)

2.1.3.6 Uncertainties in emission

The uncertainties in emissions from CROB were analyzed using the Monte Carlo technique (e.g., Ni *et al.*, 2015; Zhang *et al.*, 2019; Zhao *et al.*, 2011; Zhou *et al.*, 2017). In a few papers, the coefficient of variation (CV) for an activity data for open burning of crop residue was considered to be 20% (Ni *et al.*, 2015; Zhou *et al.*, 2017). Considering it, the uncertainty of residue burned was set at 20%. EFs are the most important component in determining the uncertainty of emissions (Zhang *et al.*, 2019). Country-based EFs were given first consideration when selecting the EFs. When the country-level value was not reported, EFs of the region and world were taken instead. The literature on regional and global EFs is inconsistent. As a result, the mean and SD of the obtained data sets were determined (Fig. 2.1). Averaging 20,000 Monte Carlo simulations with a 95% confidence interval transformed to a percentage of the mean was used to quantify the range of agricultural residue burned, EFs, and emissions (Appendix I).

2.1.4 Results and discussion

2.1.4.1 Dry matter generation

From 2003/04-2016/17, the average trend in dry matter (DM) generation has risen (Fig. 2.3). In 2016/17, it was estimated as 13,500 Gg, 29% higher than in 2003/04. In 2016/17, rice straws generated 6,670 Gg of DM, with wheat straws (2,340 Gg), maize straws (1,840 Gg), residue from sugarcane (854 Gg), residue from potato (606 Gg), residue from pulses (460 Gg), residue from oil crop (343 Gg), residue from millet (294 Gg), residue from barley (32.9 Gg), and residue from jute (11.3 Gg) (Appendix II).

2.1.4.2 Fuel use, animal feed and thatching of house

The amount of agricultural waste used as fuel has increased on average (Fig. 2.3). In 2016/17, it was estimated to be 1,410 Gg, 29% higher than 2003/04. Of the entire dry matter generated, 76% of it was utilized as animal feed. The trend of average DM consumption has risen, reaching a new high in 2016/17 (i.e., 10,200 Gg). In 2016/17, the use of animal feed has increased across the country, from 8,820 Gg in 2003/04 to 10,200 Gg (i.e., by 16%). Similarly, the usage of dry residue for thatching of roof has increased from 114 Gg for 2003/04 - 133 Gg for 2016/17 (i.e., a 17% rise) (Das *et al.*, 2020).

2.1.4.3 Crop residue open burning

By deducting the overall pattern of residue consumption from the total amount of residue generation at each district level, the quantity of agricultural residue available for burning was estimated. In the districts where there was a shortfall in the consumption of residue, open burning was predicted to be negligible. The quantity of residue to be burnt openly was estimated by summing the excess value of consumption patterns of crop residue by district level. Annual trends in residue burning have risen (Fig. 2.3 & Appendix IV). In 2003/04, the quantity of crop residue burnt openly was estimated as 2,280 Gg, somewhat greater than the 2,000 Gg stated by the literature (Streets *et al.*, 2003). Likewise, in 2016/17, it was calculated as 2,908 Gg (61-139%, i.e., having uncertainty in the range of 1,770-4,040 Gg), 28% higher than in 2003/04 (Das *et al.*, 2020). However, it indicates a substantial decline in the year 2015/16 that might be attributable to the devastating earthquake in Nepal in the month of April of year 2015. Regular farming activities may have been substantially disturbed during this emergency time as a result of the extensive damage caused by this disaster, resulting in a reduction in open burning. The percentage of residue burnt in the agricultural field was calculated as 25% in 2016/17. In 2013/14, CROB was recorded as highest, with a 27% of residue burned (Das *et al.*, 2020).

In this study, the proportion of agricultural residue burned openly was compared to other published reports in Nepal. According to NAFLQML (2019), 30% of the straw in the Tarai region is burned openly as a result of the use of combine harvesters. A default value of 25% was given for South Asia in another study (Streets et al., 2003). Furthermore, for India (Sahai *et al.*, 2010) and Pakistan (Irfan *et al.*, 2014), same value was used. According to the report, CROB trends are influenced by generation of residue and its utilization patterns (Fig. 2.3).

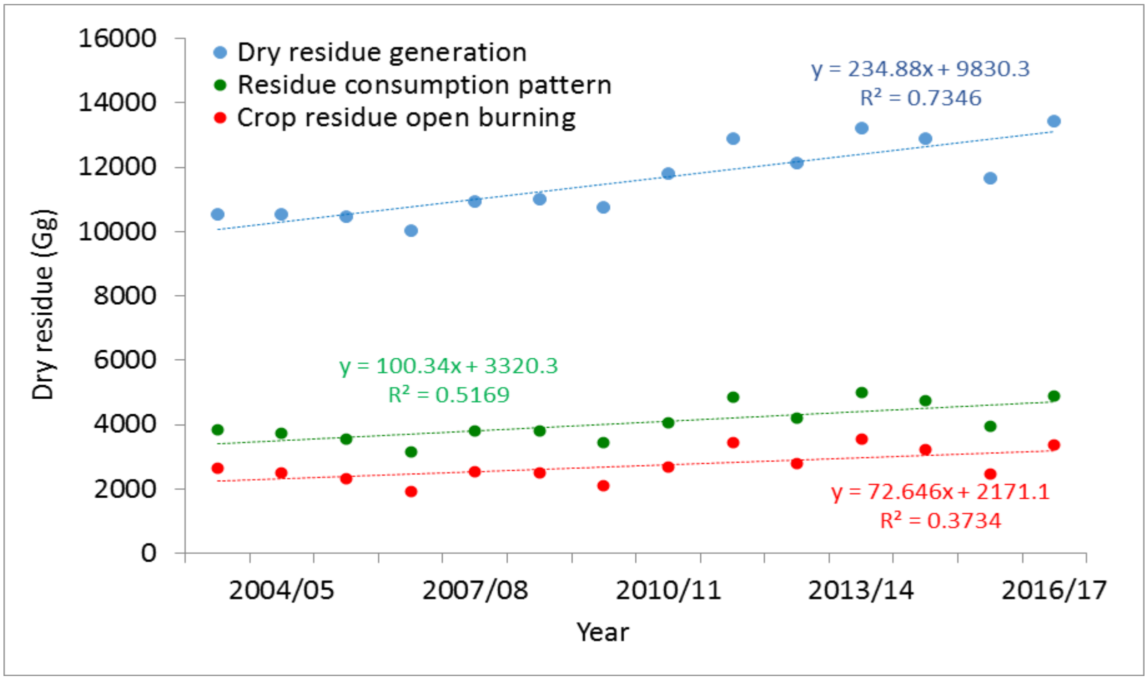

Figure 2.3: National trends in residue generation, consumption pattern and open burning (2003-2017)

2.1.4.4 Emissions from crop residue burning

The national air pollutant emissions from CROB are presented in this study for the years 2003/04 to 2016/17. The findings point to an upward trend in yearly emissions (Fig. 2.4 & Appendix V). It's worth noting that the highest emissions occurred in 2013/14, while the lowest occurred in 2006/07. In the year 2016/17, GHG emissions were calculated as CO_2 4,140 Gg (uncertainties ranges in between 56-144%) and CH_4 6.5 Gg (7-193%). The particulates pollutants were $PM_{2.5}$ 24.5 Gg (30-170%), OC 8.6 Gg (38-162%), BC 2.2 Gg (-1-201%). Likewise, trace gaseous were CO 154 (4-196%), SO_2 1.2 (60-140%), NO_x 7 (54-146%), NMVOC 22.5 (8-192%) and NH_3 2.7 (3-197%), in Gg (Table 2.5-2.6). All pollutants in the Fig. 2.4 increased by 27%, a significant rise from 2003/04-2016/17 (Das *et al.*, 2020). The years with the highest emissions were 2013/14 and the years with the lowest were 2006/07. In Appendix VI, the pollutants emission by district are shown.

This study's open burning emissions were compared with the emission inventory of the technology-based sector for the year 2011 in Nepal (Sadavarte *et al.*, 2019) (Table 2.5). According to the findings, CO_2 emissions were 2.1 folds, CO 11.1 folds, CH_4 16.9 folds,

SO_2 20 folds, OC 9.7 folds, $PM_{2.5}$ 8 folds, BC 10.5 folds, NO_x 9.1 folds, and NMVOC 18.1 folds lower than the study. These values were compared with Chinese context (Li *et al.*, 2017; Ni *et al.*, 2015; Zhang *et al.*, 2019; Zhou *et al.*, 2017); Indian context (Pandey *et al.*, 2014); and South East Asian and Central Asian contexts (Pandey *et al.*, 2014); and South East Asian and Central Asian contexts (GFED, 2020). Compared to China and India, Nepal has a substantially lower contribution of CROB to emissions, according to our findings. The share percentages of CO_2 in relation to South East Asia (GFED, 2020) was 10.4%; CO 5.9%; CH_4 4.3%; SO_2 12%; OC 17.2%; $PM_{2.5}$ 15.3%; BC 11.6%; NO_X 8.8%; and NH_3 4.8%. Likewise, the share of CO_2 in relation to the worldwide context (GFED, 2020) was 1.2%, CO 0.7%; CH_4 0.5%; SO_2 1.3%; OC 1.6%; $PM_{2.5}$ 1.7%; BC 1.3%; NO_x 1%; and NH_3 0.5% (Table 2.6) (Das *et al.*, 2020).

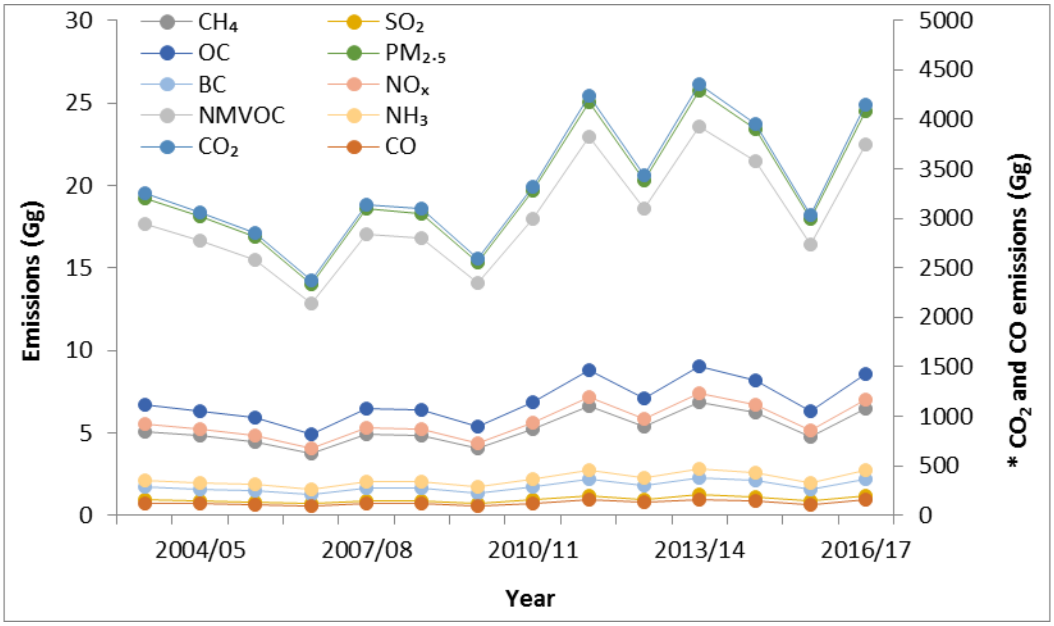

Figure 2.4: National emission trends from open burning of crop residue (2003-2017)

Table 2.5: Comparative estimates of emissions from technology-based sources in Nepal (in Gg).

	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
Technology-based emissions (2011)[a]	8900	1714	110	24	83	195	23	64	407	-
Crop residue open burning (2016/17)[b]	4140	154	6.5	1.2	8.6	24.5	2.2	7	22.5	2.7

[a] Sadavarte *et al.*, 2019; [b] This study

* CO_2 emission is from the combustion in the agricultural field and doesn't represent net emissions. Also, note that technology-based emission inventory includes sectors like residential, industry, commercial, transport, and agriculture (pumps, power tillers, tractors and thresher).

Table 2.6: Comparative estimates of emissions from global perspective.

		CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
Nepal(2016/17)[a]	Gg	4140	154	6.5	1.2	8.6	24.5	2.2	7	22.5	2.7
	kg/capita/yr	*156.26*	*5.81*	*0.25*	*0.05*	*0.32*	*0.92*	*0.08*	*0.26*	*0.85*	*0.10*
China (2008)[b]	Gg	120000	4600	-	-	390	880	-	-	-	-
	kg/capita/yr	*90.4*	*3.5*	-	-	*0.3*	*0.7*	-	-	-	-
China-in avg (2012)[c]	Gg	-	-	-	-	426	990	-	-	-	-
	kg/capita/yr	-	-	-	-	0.3	0.7	-	-	-	-
China (2012)[d]	Gg	207316.8	7563.6	598.4	83.5	478.2	1234.3		426	1282	86.3
	kg/capita/yr	*153.11*	*5.59*	*0.44*	*0.06*	*0.35*	*0.91*	-	*0.31*	*0.95*	*0.06*
China (2014)[e]	Gg	305200	18330	1070	160	710	1770	130	530	1850	120
	kg/capita/yr	*223.13*	*13.40*	*0.78*	*0.12*	*0.52*	*1.29*	*0.10*	*0.39*	*1.35*	*0.09*
India (2010)[f]	Gg	-	11356	470	74	359	1020	80	448	1745	-
	kg/capita/yr	-	*9.38*	*0.39*	*0.06*	*0.30*	*0.84*	*0.07*	*0.37*	*1.44*	-
South East Asia (2016)[g]	Gg	40000	2600	150	10	50	160	19	80	-	56
	kg/capita/yr	*62.42*	*4.06*	*0.23*	*0.02*	*0.08*	*0.25*	*0.03*	*0.12*	-	*0.09*
Central Asia (2016)[g]	Gg	60,000.00	4100	230	16	90	250	30	120	-	88
	kg/capita/yr	*873.36*	*59.68*	*3.35*	*0.23*	*1.31*	*3.64*	*0.44*	*1.75*	-	*1.28*
World (2016)[g]	Gg	360000	23700	1350	93	530	1450	174	720	-	504
	kg/capita/yr	*48.53*	*3.19*	*0.18*	*0.01*	*0.07*	*0.20*	*0.02*	*0.10*	-	*0.07*

[a] Estimated from this study
[b] Ni *et al.* (2015)
[c] Li *et al.* (2017)
[d] Zhou *et al.* (2017)
[e] Zhang *et al.* (2019)
[f] Pandey *et al.* (2014)
[g] GFED (2020)

Per capita emissions for each country was calculated referring:

Population of Nepal in 2016: CBS (2017); Population of China in 2008, 2012, and 2014 (China Statistical Yearbook, 2015); Population of India in 2016 (Census of India, 2019); Population of South East Asia in 2010, projected to 2016 based on available population growth rate (Jones, 2013); Population of Central Asia in 2016 (Policy Contribution, 2017); and Population of the World in 2016 (Population Reference Bureau, 2016).

2.1.4.5 Spatio-temporal variations and BC flux concentration

Depending on the geographical location of Nepal, open burning of crop residue varies. In 2016/17, the Tarai region had the highest percentage of residue burning (~ 91%), followed by the hilly region (~ 6%), and the mountain region (~ 3%). There could be a number of reasons for it. The Tarai region, with its widely accessible land-use area, has a larger capability for agricultural production (hence, 66.8% residue generation too) than the hilly region (27.7%) and the mountain region (5.5%) (Fig. 2.5) (Das *et al.*, 2020).

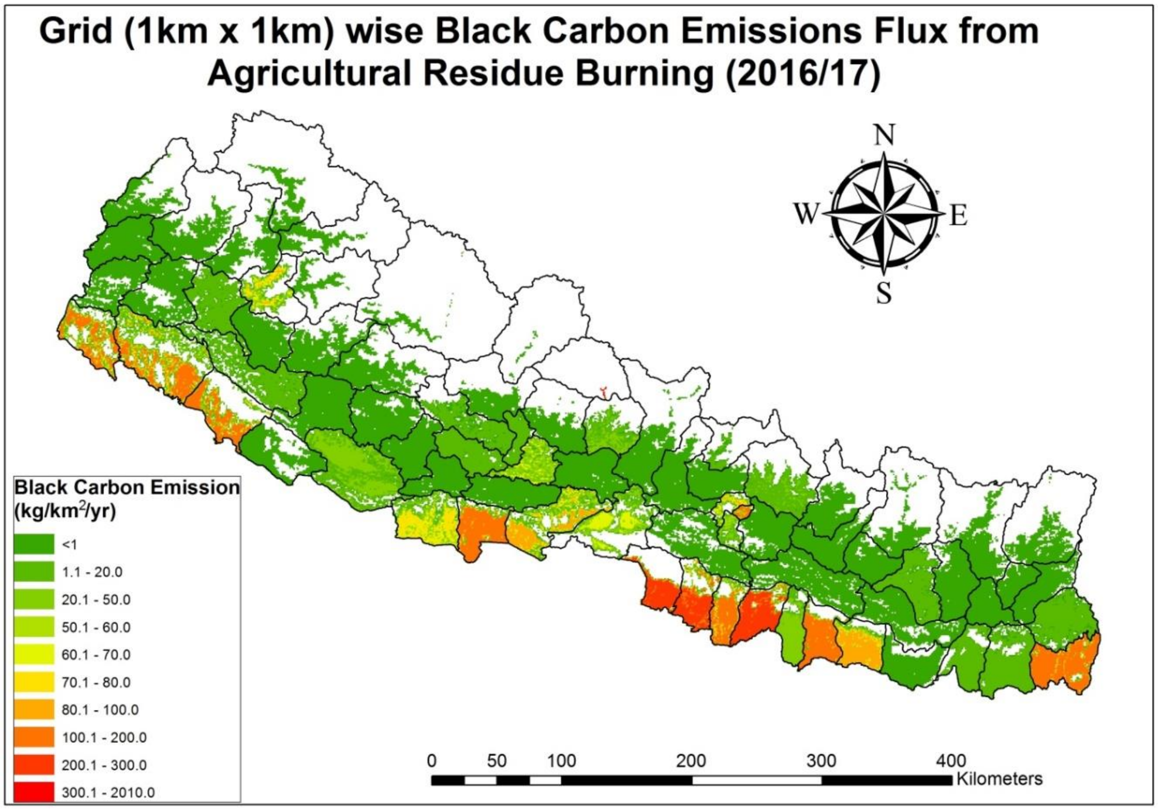

Figure 2.5: Emission flux of BC from open burning of crop residue for the year 2016/17.

Note: The emissions are negligible in the districts like Arghakhanchi, Bajhang, Baitadi, Banke, Bajura, Bhojpur, Dailekh, Dadeldhura, Darchula, Dhankuta, Dhading, Dolakha, Doti, Dolpa, Gorkha, Humla, Jumla, Jajarkot, Kavre, Kaski, Makwanpur, Mustang, Mugu, Myagdi, Nuwakot, Okhaldhunga, Panchthar, Palpa, Pyuthan, Rolpa, Ramechap, Sankhuwashava, Salyan, Sindhuli, Saptari, Solukhumbu, Taplejung, Terhathum, Tanahu, and Udayapur.

Ten different colors are used to emphasize the emission flux. Dark green to dark red indicates the lowest emission flux to the highest emission flux. Each district's white zone reflects an area where there was no emission flux. The area that is dark green is where residual burning is thought to be insignificant.

Furthermore, CROB was governed by the availability and demand for dry feed. Residue burning was low to non-existent in regions where there was a feed shortfall, according to this study. This also resulted in low to zero emissions, especially in the mountain and hilly region. Residue burning was prevalent in places where feed was plentiful, resulting in higher emissions (Fig. 2.5 & Appendix VI). According to studies, increased use of combine harvesters in recent years has also been associated with higher residue burning (Gupta, 2014; Val-Aguasca *et al.*, 2019; Yang *et al.*, 2008). According to the literature, there has been a rise in combine harvesters in Nepal's Tarai region since 2000 (Mandal *et al.*, 2017; NAFLQML, 2019; Takeshima, 2017). The rice-wheat system reported by Gupta *et al.* (2004) and labour migration are the causes of the rise in mechanization. According to this study, there is a modest association (R^2 = 0.3002) with open burning and country GDP or remittances as a result of labor migration (World Bank, 2019) between 2003 and 2017 (Fig. 2.6). According to the available study, a labor scarcity and high pay result in more open burning (Gupta *et al.*, 2004; Gupta, 2014; Jethva *et al.*, 2019). Farmers purposely burn crop residue for pest and disease control and nutrient release in the agricultural field to improve production during the next cycle, according to other research (Chang *et al.*, 2013; Gadde *et al.*, 2009; Kadam *et al.*, 2000).

The gridded inventory (1 km x 1 km) of BC emissions for the district-level for Nepal in cultivated areas is shown in Figure 2.5. The Tarai region had the highest BC emissions, whereas the mountains and hills had the lowest. In addition to carbon dioxide, BC is the second most significant anthropogenic climate force, which is caused by incomplete combustion (Bond *et al.*, 2013). It significantly melted glacier of Hindu Kush Himalayan (Bond *et al.*, 2013; Yang *et al.*, 2020; Yi *et al.*, 2019) and severely affected health (Highwood and Kinnersley, 2006). This study likewise indicated greater temporal emissions (86.16% of total emissions) from February to May, with a peak in April (Fig. 2.7-2.8 & Appendix VII). Open burning practices in sizes and volumes measurable by satellite reduce or halt entirely with the advent of the monsoon rains. Therefore, emissions were negligible in the monsoon period.

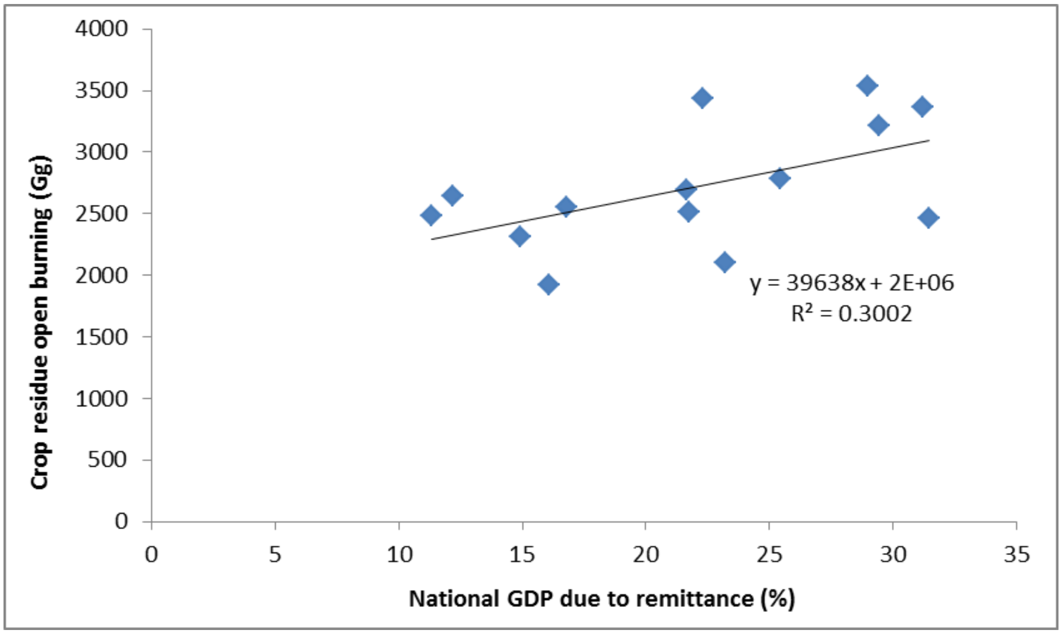

Figure 2.6: Association in between open burning of crop residue and country's GDP due to remittance/labour migration.

Note that national GDP was reported for calendar years 2003 to 2016 (World Bank, 2019). During calculation, this study assumed them as 2003/04 to 2016/17, respectively.

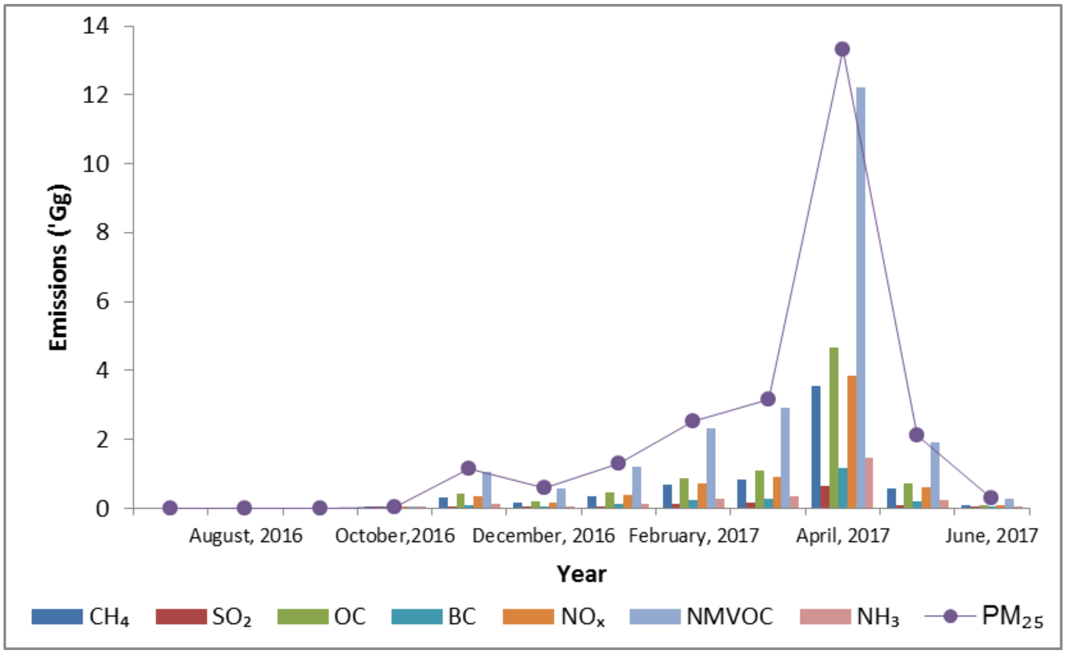

Figure 2.7: Monthly variation of emissions (2016/17).

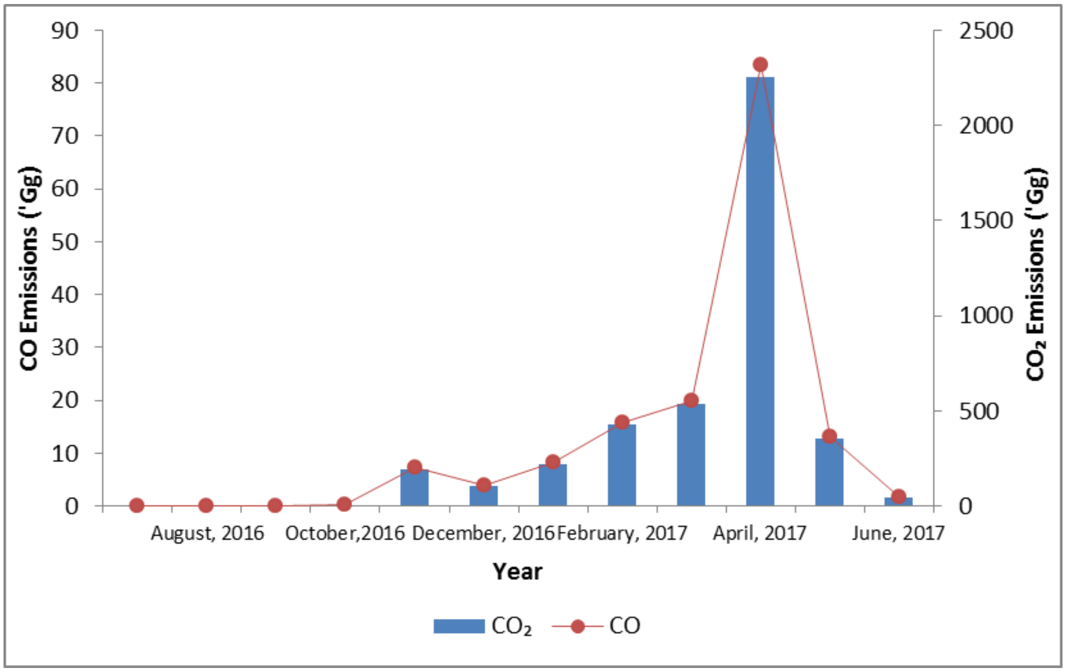

Figure 2.8: Monthly variation of CO_2 and CO emission (2016/17).

2.1.4.6 Management opportunity of air quality

In comparison to the hills and mountains, the Tarai region is more susceptible to air pollution from CROB. Detrimental pollutants that result from CROB can cause major health problems such as cardiovascular disease, respiratory disease, allergic hypersensitivity, and early deaths (Zhang *et al.*, 2017). Furthermore, because CO_2, CH_4, N_2O and BC are global warming potential pollutants, it can have an impact on the climate (Li *et al.*, 2017). In areas where farmers have more cattle per hectare, open burning of crop waste is less prevalent (Gupta, 2014). As a result, one of the potential mitigating approaches to minimize residual open burning is for farmers to raise animals for additional revenue. According to the previous studies (Gupta, 2014; Gupta *et al.*, 2004), combine harvesters can be modified to gather agricultural residue separately when harvesting crops. Animal fodder, production of clean energy (e.g., bio-char/ bio-briquette), and raw materials for industries (e.g., paper production, brick kilns, mushroom cultivation, etc.) may all benefit from the chopped crop leftovers. In addition, reaper harvesting technology may be used to gather leftover residue from the agricultural field. For commercial farming, a combine harvester can be modified with a residue spreader to scatter loose agricultural residue uniformly so that wheat can be grown through happy seeding method (Gupta, 2014). Air pollution emissions from CROB were distributed unevenly over the course of the year, according to a temporal study. From February to May, more than 80% of air pollutants occurred, with April being the peak month (Fig. 2.7-2.8). It is important to take note of this and should be taken into account by policymakers when managing air quality.

2.2 Municipal Solid Waste Open Burning

2.2.1 Introduction

Solid waste management (SWM) has emerged as one of Nepal's most pressing challenges. Solid waste production has increased in a similar pattern to that of the increase in urban populations. Despite this, SWM has been a low concern in many municipalities, owing to a variety of other demands (ADB, 2013). The various factors, such as population growth, rapid expansion of sprawling urban municipalities, growing levels of industrial and commercial activity, and rising consumption of packaged products have all contributed in severe air and water quality problems, poor sanitation, and disease transmission (Alam *et al.*, 2008; Dangi, 2009; Pokhrel and Viraraghavan, 2005). Furthermore, tardy garbage management and filthy disposal methods are causing severe environmental deterioration as well as public health problems in many municipalities (Alam *et al.*, 2008; Nagpure *et al.*, 2015). In Nepal, irregular waste collection results to open burning, which is becoming more widespread. To deal with the issues and challenges and for the better management of solid waste, Solid Waste Management National Policy (1996), Solid Waste Management Act (2011) as well as Solid Waste Management Rules (2013) were developed in Nepal. All these policy instruments emphasizes the importance of minimizing waste sources, re-use, processing, or discharge, recycling, treatment and preserving a clean and healthy environment by modifying and unifying legislation relating to solid waste management in order to decrease adverse effects on public health as well as environment. Despite have these policy tools, deeper insights of MSW open burning is not considered. Open burning of MSW is also becoming more well-known in cities in other developing countries according to Guttikunda *et al.* (2014), Hodzic *et al.* (2012), Nagpure *et al.* (2015), and Weidinmyer *et al.* (2014). In Nepal, instead of using a suitable incineration method, open burning is practiced in open areas.

Open burning of MSW is a major contributing factor for increase in emissions like particulate matter ($PM_{2.5}$, PM_{10}, BC, OC), greenhouse gases (CH_4, CO_2), and traditional air pollutants (SO_2, NO_x, NMVOC, CO, NH_3) (Lemieux *et al.*, 2004; Shrestha, 2018). It has a significant effect on human health. The effects of prolonged exposure to PM can include cardiovascular and respiratory issues, as well as cancer and adverse birth outcomes (McDonnell *et al.*, 2000). Pollutants produced by MSW open burning, such as carbon dioxide (CO_2) and black carbon (BC), have a large global warming potential, causing local and regional warming. When MSW is dumped, it slowly degrades, releasing methane (CH_4), another strong greenhouse gas. Open burning of MSW emits ammonia (NH_3), which can contribute to PM production in the atmosphere.

There are just a few studies on the emissions inventory from MSW open burning across the world. More than ten years ago, Gautam (2006) provided an estimate of MSW open burning that was released as the final report on an action program for managing air quality in the Kathmandu Valley. Despite the fact that MSW open burning was quantified, the literature could not highlight the methodology to be used in this study. Similarly, through the NAMaSTE initiative (Stockwell *et al.*, 2016; Jayarathne *et al.*, 2018) country-specific EFs on waste burning was generated. The emission inventory for open burning of MSW in Nepal was developed using the EFs from this literature.

Another study on the effect of waste burning on air quality of Mexico City was conducted by Hodzic, *et al.* (2012). Likewise, Lal et al. (2016) looked into how burning dung cake and MSW in Agra, India, affected the Taj Mahal and people's health. Nagpure *et al.* (2015) undertook research to analyze the geographical and temporal patterns of MSW open burning in Indian towns. Ramaswami *et al.* (2016) have conducted research on the social as well as infrastructural elements that influence MSW open burning in three Delhi areas. Kumar *et al.* (2018) conducted a research employing metal tracers and lead isotopic composition in order to understand the impact of open-waste burning on urban pollutants. Park *et al.* (2013) investigated the discharge of hazardous air pollutants in a Korean metropolitan region from open burning of residential MSW. Wang *et al.* (2017) also investigated typical harmful heavy metals from open burning of MSW in China. Wiedinmyer *et al.* (2014) conducted a research on worldwide emissions of particulate matter, hazardous air pollutants, and trace gases from household trash open burning. Those study provided insight into the methodologies as well as findings with which to compare the outcomes.

A recent study in South Asia examined the importance of rubbish burning to air quality of the region (Saikawa *et al.*, 2020). The emission pattern as well as source contribution of harmful air pollutants from MSW open burning in China were examined by Cheng *et al.* (2020). Korai *et al.* (2020) conducted a study that compared MSW management practice in Pakistan and China. According to the literature, measuring emissions from open burning of MSW is either totally ignored (Pradhan *et al.*, 2012) or done extremely poorly (Gautam, 2006; Wiedinmyer *et al.*, 2014). The difficulty and significant uncertainty in determining the proportion of MSW that is actually burnt are the reasons behind this (Nagpure *et al.*, 2015). Unlike other studies, this one takes into account activity parameters such as fraction of population that burns waste, fraction of waste that is combustible and spatial distribution of waste-burning. Moreover, country-specific EFs were given the highest priority. This study aims to resolve some of these uncertainties by developing a reliable MSW open burning emission inventory for the Kathmandu Valley and Nepal.

2.2.2 Objectives

The general objective of this study was to develop emission inventory from MSW open burning in the context of Nepal.

The specific objectives were:

I. To estimate MSW generation of Nepal based on secondary sources of information.
II. To estimate waste combustible fraction of MSW.
III. To estimate fraction of population burning MSW at sources and disposal sites of Kathmandu Valley and Nepal.
IV. To estimate MSW open burning and emissions for Kathmandu Valley and Nepal.
V. To spatially allocate emission flux from MSW open burning at 1 km x 1 km fine resolution.

2.2.3 Materials and methods

2.2.3.1 Population estimation

The population of each district and the urban population in 2001 and 2011 were taken from CBS (2012). From its information, compound annual growth rate (CAGR) was calculated referring eq. 2.5 (Sivaprasad, 2012). The CAGR is the population's average yearly growth rate during a period of time longer than one year. For all districts and municipalities, the CAGR (in fraction) was used to compute the population up to 2016. By subtracting the municipality population (in number) from the district population (in number), each district's rural population (in number) from 2001-2016 was calculated.

$$\mathrm{CAGR} = \left[\left(\frac{\text{Final Population}}{\text{Initial Population}}\right)^{\frac{1}{\text{Years}}}\right]-1 \quad (2.5)$$

A population factor (2001-2016) was further calculated for each municipality and rural district in Nepal, which was used in estimating the fraction of population burning MSW from 2001-2016.

2.2.3.2 Municipal solid waste generation

Nationally a plethora of studies on municipal solid waste generation rate (MSWGR) were carried out in the municipality level only. The most widely used studies are ADB (2013), Alam *et al.* (2008), Dangi *et al.* (2011), Nippon Koei Co., Ltd and Yachiyo Engineering Co., Ltd. (2005), SWMRMC (2008), and SWMTSC (2015). For the Kathmandu Valley, ADB (2013) and SWMTSC (2015) were considered because they are the latest studies and contain information on all municipalities that were studied. Likewise, at the national level, the MSWGR for municipalities for 2012 was collected from ADB (2013) and for 2003 from SWMRMC (2004). The MSWGR of 2003 was further validated and updated with other study (NIPPON KOEI CO., LTD., YACHIYO ENGINEERING CO., LTD., 2005). The MSWGR for rural area is not available in any study yet, so we considered a recent study of 60 new municipalities (SWMTSC, 2017).

Before the new constitution (September, 2015) in Nepal, these newly formed municipalities were considered rural villages. Moreover, the settlement settings are still of the semi-rural type. The MSWGR from the remaining 15 rural districts was assumed to have the same value as the nearby rural district. After multiplying MSWGR (kg/person/day) by the population (in number) of each municipality and rural district, the quantity of MSW (in kg) was calculated (Eq. 2.6).

$$\mathrm{MSW} = \mathrm{MSWGR} \times \mathrm{Population} \qquad (2.6)$$

CAGR was further applied to estimate MSW from 2003 to 2016 for municipalities. Referring to years 2003 and 2016 (i.e., municipalities) and 2016 (rural districts), the MSW value of 2003 for rural districts was calculated (Eq. 2.7). CAGR was applied to estimate MSW from 2003 to 2016 for rural districts.

$$\mathrm{MSW}_{\mathrm{Rural\ district\ (2003)}} = \frac{\mathrm{MSW}_{\mathrm{Municipalities\ (2003)}} \times \mathrm{MSW}_{\mathrm{Rural\ district\ (2016)}}}{\mathrm{MSW}_{\mathrm{Municipalities\ (2016)}}} \qquad (2.7)$$

2.2.3.3 Field study in the Kathmandu Valley

Urban core and sub-urban areas were identified in KMC, LSMC, and Bhaktapur. The research routes used market centers in both the metropolitan core and the suburbs as reference points. The study routes were defined as the primary routes that intersected with the market center. These were the routes used for the transect walk investigation.

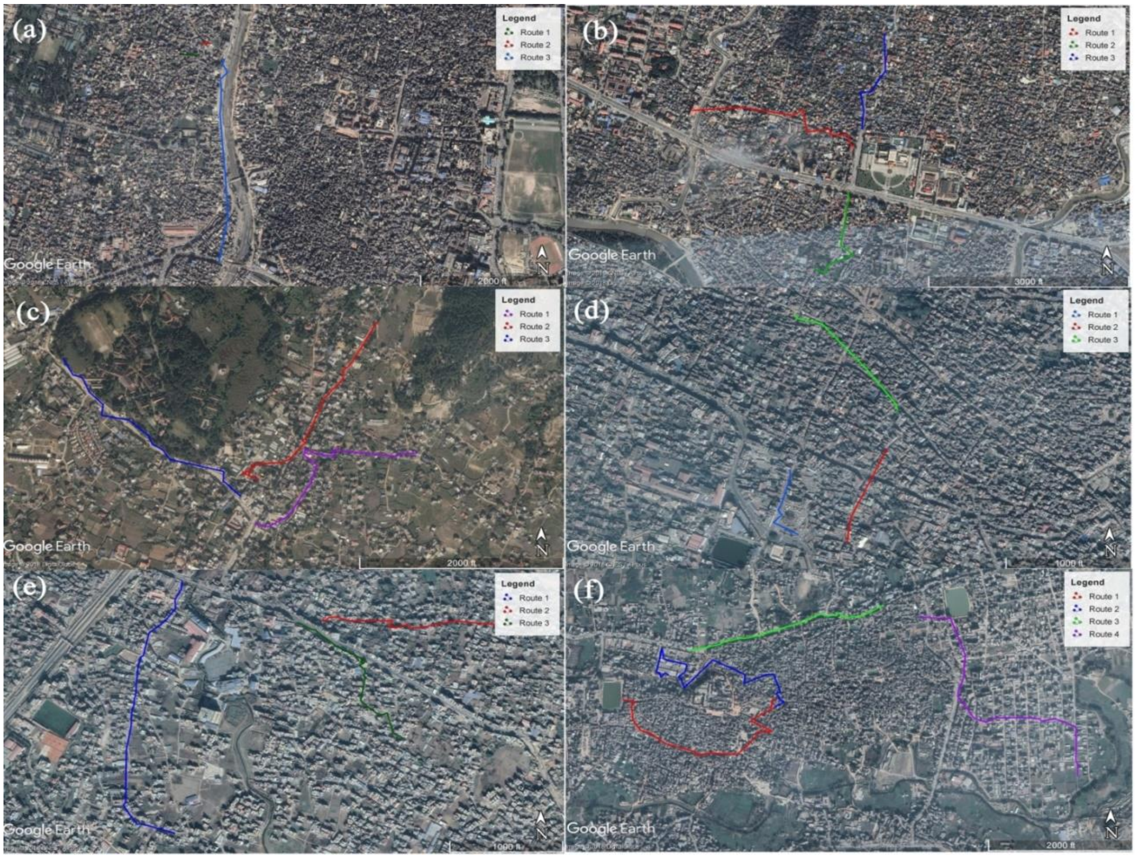

Figure 2.9: Study routes (Google Earth)

(a) Kalimati/Dallu (urban core); (b) Baneshwor (urban core); (c) Budhanilkantha (sub-urban); (d) Lagankhel (urban core); (e) Mahalaxmi/Gwarko (sub-urban); (f) Bhaktapur (urban core and sub-urban)

A field study was conducted from April to June 2016 in seven selected portions of KMC, including Budhanilkantha, former LSMC, and Bhaktapur municipality. The study areas were Kalimati/Dallu (core) and Baneshwor (core) of KMC; Budhanilkantha (sub-urban); Lagankhel (core) and Mahalaxmi/Gwarko (sub-urban) of former LSMC; and Sukuldhoka/Durbar Square (core) and Kamalbinayak/Chyamasingha (sub-urban) of

Bhaktapur municipality. Within these seven study areas, transect walks and household surveys were carried out along a total of 19 different routes (Fig. 2.9). At the same time that the trash mounds were being counted along the transect line/study route. The sample size was determined based on the number of routes, household density, and the size and length of each route in the urban core and sub-urban areas. In sparsely inhabited areas, every fifth household (i.e., 1, 6, 11, etc.) was surveyed, while in more densely populated areas, every tenth household was surveyed. After taking into account the full investigation, 179 houses were interviewed.

2.2.3.4 Estimation of the waste combustible fraction

To calculate waste combustible fraction (δ), a field experiment was conducted in Kirtipur near Tribhuvan University (Fig. 2.10). The following were the steps in the experiment: A research was first carried out to determine the trash composition of each pile. Second, a digital weighing balance was used to determine initial mass of MSW. Third, the experimental garbage piles were burned carefully until it converted to ash. Finally, the remaining trash and ash were carefully separated and their masses were measured individually. By subtracting the mass of leftover trash after combustion from the total initial mass, the burnt mass of waste was calculated.

Waste combustible fraction is the ratio of burned quantity of MSW to total initial quantity of MSW (IPCC, 2006).

$$\delta = \frac{\text{Burnt Mass of Trash}}{\text{Total initial mass of Trash}} \qquad (2.8)$$

The open burning of MSW was calculated using this experimentally obtained parameters. The amount of waste combustion is determined by a number of parameters, including waste type, moisture content, and weather conditions (e.g., precipitation, temperature, wind). It also differs depending on the source and the landfill/dumping location.

Figure 2.10: Waste combustion experiment

2.2.3.5 Estimation of the fraction of population burning MSW

During the field study's transect walk, the total number of garbage piles and the incidence of burning were identified and noted down. Household surveys were conducted concurrently to get information on total population residing in waste piles burning routes. In addition, data on active participation in waste collection services was gathered. Information on waste management practices (e.g., burying, burning, and composting) was acquired during interviews for persons who did not participate at garbage collecting facilities. The household perception data on waste burning was verified using the following information. The Fraction of population burning MSW (P_{frac}) was then estimated, which is ratio of population who is not participating at waste collection to population whose waste is collected for disposal or landfilling (IPCC, 2006). It was anticipated that those who did not actively participate in garbage collection services would burn their trash.

2.2.3.6 Estimation of the fraction of MSW burning at disposal sites

The waste collection vehicles were followed from waste collection points to disposal points in all the study routes to know the status of waste that was really dumped or burnt at the disposal site (λ). The study team tracked each waste collection point before 5:00 am in the morning. Motorbikes were used to track the vehicles. Different stakeholders such as Department of Environment, KMC, Teku; Department of Environment, LSMC, Balkumari; Bhaktapur municipality; and locals) were consulted before conducting this study to determine the availability of drivers, waste collection vehicle types, routes, time, and frequency of waste collection.

2.2.3.7 Waste burning parameters for national estimate

In the estimation of MSW open burning in Nepal from 2001 to 2016, waste burning parameters "such as, fraction of population burning MSW (P_{frac}); waste combustible fraction (δ); fraction of MSW burning at disposal sites (λ); and burning/oxidation efficiency (η)" were used. The findings from a field study in the Kathmandu Valley were used to estimate P_{frac} for municipalities and rural districts. The municipality and rural district of P_{frac} are not available in the literature. To estimate garbage burning parameters, this study differentiated the local administrative divisions (58 municipalities and 75 rural districts). The P_{frac} values of 53 municipalities were assumed to be the same as those of Bhaktapur. Similarly, the P_{frac} of four sub-metropolitan cities was assumed to be the same as LSMC's, and KMC had the real-scenario value. During the time of the study, Budhanilkantha area was completely under a village development committee. Therefore, P_{frac} for rural districts were assumed as Budhanilkantha rural area. The trend of P_{frac} was calculated by multiplying P_{frac} by the population growth factor (2001-2016).

Moreover, λ was based on field survey in the Kathmandu Valley as well as available literatures (ADB, 2013; SWMRMC, 2004) and η from Shrestha (2018). A field assessment was done in the Kathmandu Valley to establish the condition of garbage that was actually dumped or burned at the disposal site. Furthermore, various stakeholders

(Department of Environment, KMC, Teku; Department of Environment, LSMC, Balkumari; Bhaktapur municipality; and locals) were consulted on garbage collection services, time, routes, waste collection frequency, and driver availability. To confirm the presence of MSW open burning practices, visual examination and surveillance were done at disposal sites in the Kathmandu Valley. Likewise, to investigate MSW open burning practices in 58 municipalities of Nepal, ADB (2013) was considered.

2.2.3.8 Emission estimation

The emissions from MSW open burning for municipalities in the Kathmandu Valley were estimated for 2016, whereas for the country it was considered between 2001 and 2016. Various literatures were used to calculate MSW burned at municipal and rural district source and disposal sites (e.g., IPCC, 2006; Shrestha *et al.*, 2013a; Shrestha, 2018), eqns. (2.9) and (2.10).

2.2.3.8.1 Solid waste open burning at source:

$$\mathrm{M_s} = \mathrm{Pc} \times \mathrm{MSWGR} \times \delta \times \mathrm{P_{frac}} \times \eta \times 365 \qquad (2.9)$$

Where, M_s is the quantity of MSW open burned (kg/day); Pc is population (capita); MSWGR is per capita MSW generation rate (kg/capita/day); δ is fraction of combustible MSW; P_{frac} is fraction of population burning waste; and η is burning/oxidation efficiency (in fraction), which is 0.4 (Shrestha, 2018).

2.2.3.8.2 Solid waste open burning at disposal site:

$$\mathrm{M_s} = \mathrm{Pc} \times \mathrm{MSWGR} \times \varepsilon \times \lambda \times \delta \times \eta \times 365 \qquad (2.10)$$

Where, M_s is quantity of MSW open-burned (kg/day); Pc is population (capita); MSWGR is per capita MSW generation rate (kg/capita/day); ε is collection efficiency of MSW (fraction that is disposed or land filled); λ is fraction of the trash that is actually burnt in relation to the total mass of trash disposed in a disposal site; δ is fraction of combustible MSW, and η (in fraction) is burning or oxidation efficiency.

The total emissions from MSW open burning for municipalities in the Kathmandu Valley and Nepal (2001-2016) were estimated by multiplying activity data with EFs (Defra, 2009; IPCC, 2006; Shrestha *et al.*, 2013a; Shrestha, 2018), eq. (2.11). EF is expressed as grams of pollutant emitted per kilogram of MSW burned.

$$\mathrm{Em_i} = \mathrm{M_s} \times \mathrm{EF_i} \qquad (2.11)$$

Where, Em_i is emission (in kg) of pollutant i; EF_i (g/kg) is emission factor of pollutant i; M_s is quantity of MSW burnt.

The EFs were gathered from a variety of sources in the literature. The country-based EFs were given first priority. When the value of a nation was not reported, EFs from a global context were used instead. The EFs for CO, CO_2, NO_x, BC, $PM_{2.5}$, CH_4, OC, NH_3, and EC were collected from the recent on-field measurements during the NAMASTE campaign in Nepal (Jayarathne *et al.*, 2018; Stockwell *et al.*, 2016). The EF for SO_2 was obtained from a laboratory-based measurement (Akagi *et al.*, 2011), which is a global reported value. Likewise, EFs for PM_{10} and NMVOC were also based on a global context (USEPA, 1995).

2.2.3.9 Data analysis and presentation

A computational tool such as MS-Excel was used to compile and analyze the data, whereas R-Studio and Google Earth were used to present the results for municipalities in the Kathmandu Valley. To spatially allocate air pollutants at a national level, a fine-resolution gridded emissions inventory (1 km x 1 km) was developed from national outline/boundary, national district boundary, and land cover of 2010 (ICIMOD, 2013), and district-wise open burning data for 2011.

2.2.4 Results and discussion

2.2.4.1 Waste combustible fraction

The waste combustible fraction was determined experimentally in the Kathmandu Valley. The average was 0.57, with values ranging from 0.53 to 0.66 (Fig. 2.11). (Das *et al.*, 2018a). While in field study, the waste piles were estimated to be 50% organic, 30% plastic, 10% paper, 3% rubbers, 2% metals, 2% textile, 1% glass and 2% other material. The waste composition was determined through segregation and visual inspection. The sample organic waste piles were moist during the experiment, whereas the remaining fractions were dry. After the rain had stopped, the waste combustible fraction experiment was carried out. As a consequence, the outcomes were influenced by all of the aforementioned factors.

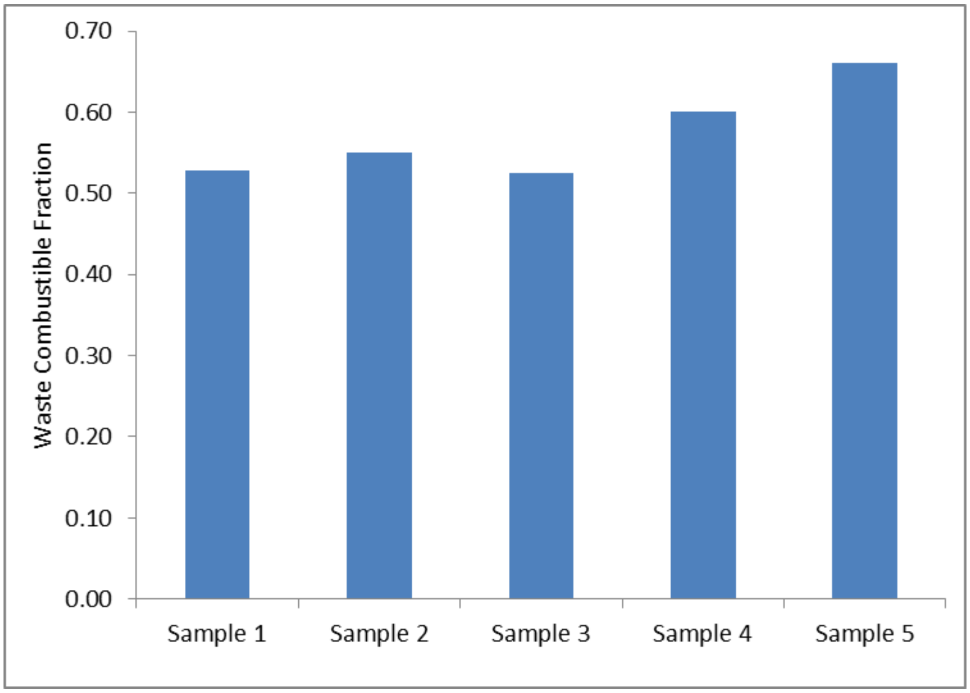

Figure 2.11: Waste combustible fraction based on field experiment, 2016

2.2.4.2 Estimation of the fraction of population burning MSW

The total proportion of MSW burned by the population (P_{frac}) was estimated to be 0.13 (Table 2.7) (Das *et al.*, 2018a). Budanilkantha's P_{frac} value was significantly greater than that of any other neighbourhood. It was caused by sporadic or late garbage collection, as well as less accessible, narrow, and sloping roadways that reached many homes. According to our findings, the higher the value of P_{frac}, the more garbage is burned or the waste collecting efficiency is low.

Table 2.7: Summary of MSW open burning in the Kathmandu Valley.

Study routes	Total weight of the trash that burnt (kg/day)	Daily per capita MSWGR (kg/capita/day)	Total daily MSW generation (kg/day)	P_{frac}	Waste burning (kg/capita/day)
Budanilkantha	21.86	0.48	383.15	0.25	0.027
Bhaktapur (core)	2.22	0.35	235.17	0.04	0.003
Bhaktapur (sub-urban)	9.13	0.35	416.73	0.1	0.008
Lagankhel	7.38	0.37	194.09	0.17	0.014
Mahalaxmi/ Gwarko	16.94	0.36	354.04	0.21	0.017
Kalimati/Dallu	4.27	0.46	348.52	0.05	0.006
Baneshwor	10.54	0.46	411.14	0.11	0.012
In overall	72.36	0.4	2342.85	0.13	0.012

2.2.4.3 Estimation of the fraction of MSW open burning at disposal sites

All the waste that was collected by vehicles was tracked all the way to the final transfer station and disposal sites. Observations were made at each of the chosen informal dumping sites, as well as ADB (2013), which was cited. At the disposal locations, no burning practices have been identified. The amount of MSW that was burnt at the disposal locations was assumed to be zero.

2.2.4.4 MSW open burning

The total mass of MSW burned in the five erstwhile municipalities (i.e., KMC, LSMC, Bhaktapur, Kirtipur, and Madhyapur Thimi) of the Kathmandu valley during 2016 is estimated to be 7,400 tons (i.e., 20 tons/day), accounting for approximately 3% of total trash generated in the valley municipalities (Table 2.8) (Das *et al.*, 2018a).

Likewise, the total quantity of MSW burnt in 2011 in Nepal is estimated as 89,000 tons (i.e., 240 tons/day), accounting for 4.5% of the total waste generation. In 2001, the total amount of MSW burned was 75,000 tons, which reached 110,000 tons in 2016, an increase of 46% in a decade. This study reveals that higher MSW open burning prevailed in rural areas than in urban areas (Fig. 2.12). In 2011, the amount of MSW burned in urban areas (58 municipalities) was 8,790 tons, and 80,000 tons in rural areas (75 rural districts). However, the trend of open burning is much more prevalent in urban than rural settings. This study estimates the increase in MSW open burning to be 4.5 folds in urban areas and 1.3 folds in rural areas from 2001-2016.

The result from this study is compared with similar South Asian cities and the world (Table 2.8). The MSW burning of valley municipalities is 131,377, 197, 48, 11, 11, 5, and 4 folds lower than global estimates, Mexico City, Mumbai, Delhi, Agra, Kanpur, and Ulaanbaatar, respectively. The amount of MSW burned in Nepal is almost similar to that in Delhi and Agra. It is higher than Kanpur and Ulaanbaatar by 2.5 folds and 2.8 folds, respectively. However, it is lower than Mumbai, Mexico City, and the world by 4 folds, 16 folds, and 10,924 folds, respectively.

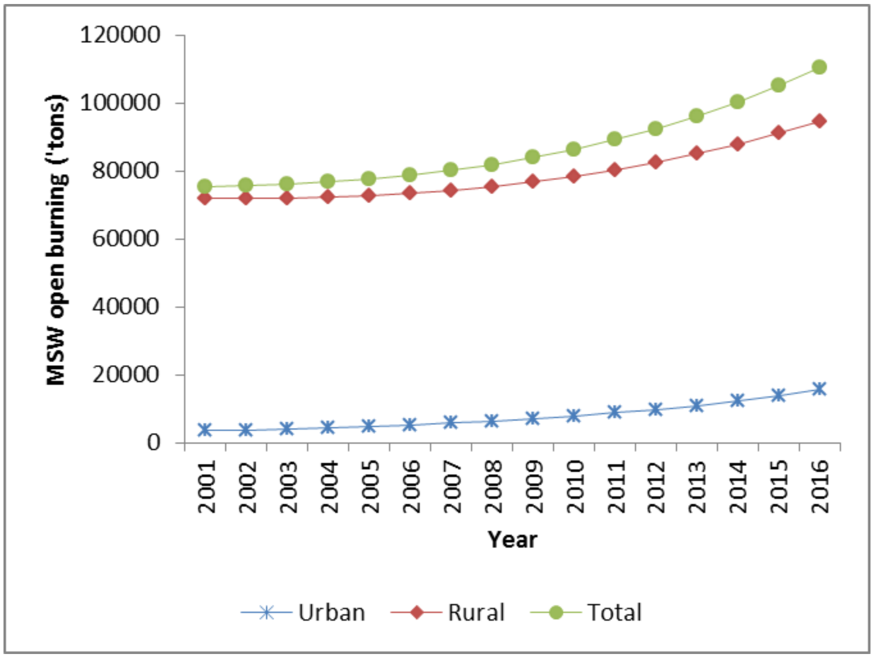

Figure 2.12: MSW open burning from 2001-2016, Nepal

The European Union and the United States estimate the MSW open burning rates of large cities at between 0.25% and 0.3% of the total waste generated (Park *et al.*, 2013). The literature suggests MSW open burning in Mexico City is high (i.e., > 50%) in the poorest areas (Hodzic *et al.*, 2012). According to Bond *et al.* (2004), Asia contributes 14 Tg/year, Africa (5 Tg/year) and the world (33 Tg/year).

This study contradicts the findings of Wiedinmyer *et al.* (2014), who estimate 972 Tg/year from residential and dump site burning. Waste burning occurs for a variety of causes and in a variety of locations. It has been noticed more frequently in locations where garbage collection services are scarce, costly, or non-existent (Wiedinmyer *et al.*, 2014). Another literature suggests garbage burning is caused by small urban areas and cities lacking landfill sites, inefficient waste collecting systems, irregular waste collection services, inexperienced disposal techniques, and a negative attitude against dump-burn activities (Guttikunda *et al.*, 2014; Ramaswami *et al.*, 2016).

Table 2.8: Comparative estimates of MSW open burning from global perspective.

Places	Waste burning (in Gg)
Nepal (2011)[x]	89
Kathmandu valley municipalities, Nepal (2016)[a]	7.4
Delhi, India (2015)[b]	79.6
Agra, India (2015)[b]	81.4
Kanpur, India (2005)[c]	35.0
Mumbai, India (2005)[d]	352.6
Ulaanbaatar, Mongolia (2005)[e]	31.5
Mexico City, Mexico (2006)[f]	1460.0
World (2014)[g]	972190

[x,a]estimation from this study; [b]Nagpure *et al.*, 2015; [c]calculated (MSW open burning percentage as of 2007 from Sharma, 2010; population as of 2005 from United Nations, 2012; MSWGR as of 2004/05 from CPCB, 2017); [d]calculated (MSW open burning percentage as of 2005 from Sharma, 2010; population as of 2005 from United Nations, 2012; MSWGR as of 2004/05 from CPCB, 2017); [e]calculated (MSW open burning percentage as of 2005 from Guttikunda, 2007; population as of 2005 from United Nations, 2012; MSWGR as of 2005, avg of summer and winter from JICA, 2007); [f]Hodzic *et al.*, 2012; [g]Wiedinmyer *et al.*, 2014
MSW open burning percentage for 2005 is assumed same as that of 2007 for Kanpur.

2.2.4.5 MSW open burning and emissions

The total emissions from trash burning from the five erstwhile municipalities of the Kathmandu Valley were estimated as $PM_{2.5}$ 55 tons (OC 42 tons and EC 1.4 tons), PM_{10} 60 tons, BC 25 tons, CO_2 11,900 tons, CH_4 30 tons, SO_2 5.0 tons, NO_x 19.2 tons, CO 630 tons, NMVOC 112 tons, and NH_3 5.7 tons in the year 2016 (Fig. 2.13) (Das *et al.*, 2018a).

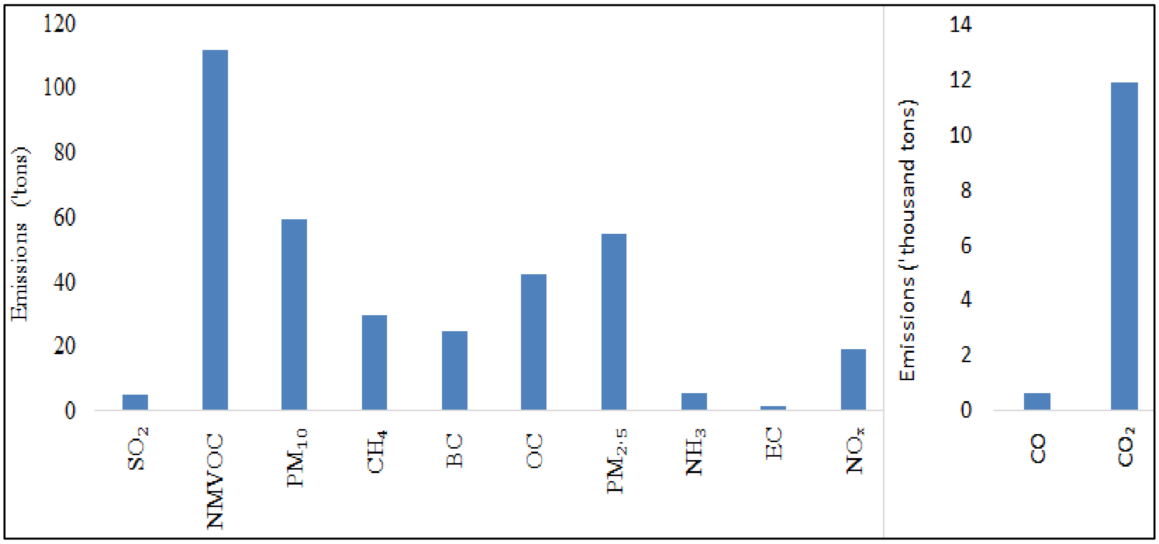

Figure 2.13: Emissions from MSW open burning, valley municipalities

Nationally, this study estimates air pollutants from MSW open burning to be $PM_{2.5}$ 667 tons (OC 514 tons), PM_{10} 724 tons, BC 299 tons, CO_2 144,900 tons, CH_4 359 tons, SO_2 60 tons, NO_x 233 tons, CO 7,665 tons, NMVOC 1,357 tons, and NH_3 69 tons in a year 2011 (Fig. 2.14-2.15). The emissions "such as, $PM_{2.5}$, OC, PM_{10}, BC, CO_2, CH_4, SO_2, NO_x, CO, NMVOC, and NH_3" increased by 46.7% from 2001 to 2016 (Appendix VIII). The national trends of emissions from 2001-2016 are presented in the figure 2.14-2.15.

The findings were compared to previous estimates from diesel generator (DG) sets, vehicle traffic, and manufacturing industries for different years in order to establish MSW open burning as one of the major sources in the Kathmandu Valley. As per World Bank (2014), DG set emissions for the Kathmandu Valley in 2012/2013 was reported.

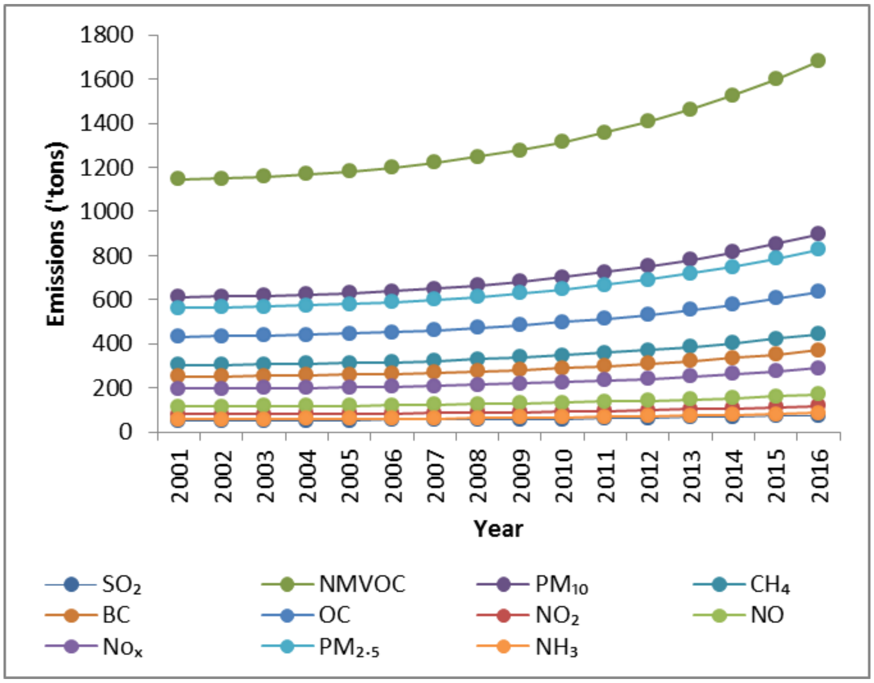

Figure 2.14: Emissions from MSW open burning from 2001-2016, Nepal

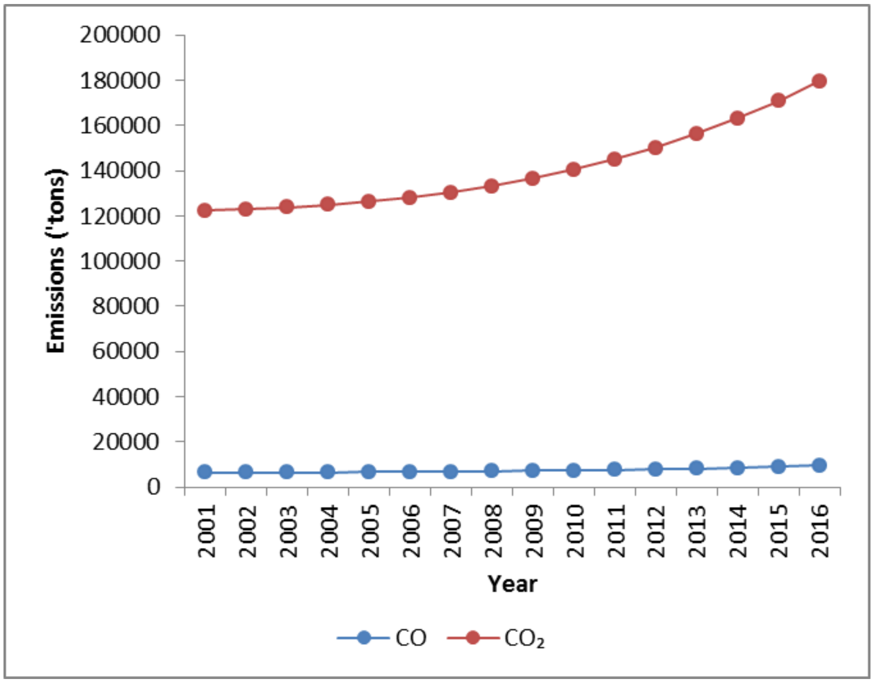

Figure 2.15: Emissions from MSW open burning from 2001-2016, Nepal

The emissions from MSW open burning for SO_2, NO_x, CO, PM_{10}, BC, OC and CO_2 were estimated to be 11, 281, 2, 6, 9, 3, and 18 times lower than DG set emissions, respectively. Compared with vehicular emissions, NO_x is 834 folds lower, CO is 49 folds lower, BC is 86 folds lower and CO_2 is 130 folds lower than reported by Shrestha *et al.* (2013b). Comparing with manufacturing industries (Pradhan *et al.*, 2012), MSW open burning emits SO_2 and NO_x, which are 607 and 44 fold lower, respectively. However, CO and NMVOC are 1.6 times and 2 times higher, respectively, than manufacturing industries (Table 2.9) (Das *et al.*, 2018a). The emissions from valley municipalities and Nepal are compared with other regional and international findings. The emissions were significantly lower than Delhi, Mexico City, Ulaanbaatar, and the world (Table 2.10) (Das *et al.*, 2018a). According to the literature, PM emissions from MSW burning account for roughly 8% in India, 22% in China, 33% in Bangladesh, and 69% in Pakistan (Wiedinmyer *et al.*, 2014). In Mumbai, garbage burning and landfill fires produce 22,000 tons of air pollution every year in the form of PM, HC, CO, NO_x, and SO_2, as well as dioxins and furans (10,000 TEQ grams per year) (Annepu, 2012). Waste burning and emissions vary for a variety of reasons. Where waste collection services are scarce, expensive, or unavailable, it occurs more frequently (Wiedinmyer *et al.*, 2014). More

MSW is burned openly in small and medium-sized communities, as well as cities without landfills and/or with just partial garbage collection services (Guttikunda *et al.*, 2014). Other reasons for MSW open burning in residential and open areas include a lack of adequate trash collection services (Ramaswami *et al.*, 2016), as well as irregular garbage collection services, inadequate disposal methods, and a poor attitude about dump-burn activities. In Nepal, appropriate trash recycling systems are trifling. In the Kathmandu valley, waste is collected three times per week on average (SWMTSC, 2015).

These emissions from MSW open burning can lead to severe health impacts like heart diseases, severe respiratory diseases, and allergic hypersensitivity (McDonnell *et al.*, 2000), including impacts on the local climate because BC and CO_2 have a high global warming potential.

Table 2.9: Comparative estimates of emissions (in tons) from different sources in the Kathmandu Valley.

Emission sources	SO_2	NO_x	CO	NMVOC	PM_{10}	CH_4	BC	OC	CO_2	$PM_{2.5}$	NH_3	EC
Solid waste burning- Kathmandu Valley municipalities (2016)[a]	5	19.2	629.9	111.5	59.5	29.5	24.5	42.2	11913	54.8	5.7	1.4
DG sets- Kathmandu Valley (2012/13)[b]	54	5400	1200	-	380	-	221	120	210150	-	-	-
Vehicular emission- Kathmandu Valley (2010)[c]	-	16000	31000	-	-	-	2100	-	1554000	-	-	-
Manufacturing industries- Kathmandu Valley (2012)[d]	3014	838	389	53	-	-	-	-	-	-	-	-

a the estimation from this study; [b]World Bank (2014); [c]Shrestha, *et al.* (2013b); [d]Pradhan, *et al.* (2012)

Table 2.10: Comparative estimates of emissions (in Giga grams) from global perspective

Emission sources	SO_2	CO	NMVOC	PM_{10}	CH_4	BC	OC	CO_2	$PM_{2.5}$	NH_3	EC
Nepal (2011)[f]	0.06	7.7	1.4	0.72	0.36	0.3	0.51	145	0.67	0.07	-
Kathmandu valley municipalities, Nepal (2016)[a]	0.01	0.63	0.11	0.06	0.03	0.02	0.04	11.91	0.05	0.01	-
Delhi, India (2010)[b]	0.40	27.80	-	7.60	-	-	-	-	5.30	-	-
Mexico city, Mexico (2012)[c]	0.73	60.59	-	-	-	-	-	-	-	1.46	-
Ulaanbaatar, Mongolia (2006)[d]	-	-	-	4.07	-	-	-	-	3.05	-	-
World (2014)[e]	486	36943	-	11569	3597	632	5123	1412592	9527	1089	-

[f,a] the estimation from this study; [b]Guttikunda and Calori, 2013; [c]Hodzic *et al.*, 2012; [d]Guttikunda, 2007; [e]Wiedinmyer *et al.*, 2014

2.2.4.6 Spatial variations and $PM_{2.5}$ emission flux

The open burning of MSW differed according to the country's geographic location. Ten distinct colors are used to highlight the $PM_{2.5}$ emission flux. Light yellow to dark red denotes the lowest emission flux to the maximum emission flux. Each district's white zone reflects the area where there was no emission flux. The districts of Kathmandu Valley had the maximum $PM_{2.5}$ emission flux in 2011, followed by Kaski, Terai Region districts, hills/mid mountains, and mountains/higher Himalayas (Fig. 2.16). It could be due to numerous factors. The P_{frac} value, population size, and MSWGR are higher in the districts of the Kathmandu Valley, resulting in higher open burning and emission flux than in other areas of Nepal. In the Kaski and Terai regions of Nepal, population size, MSWGR, and P_{frac} are higher, leading to increased emission flux.

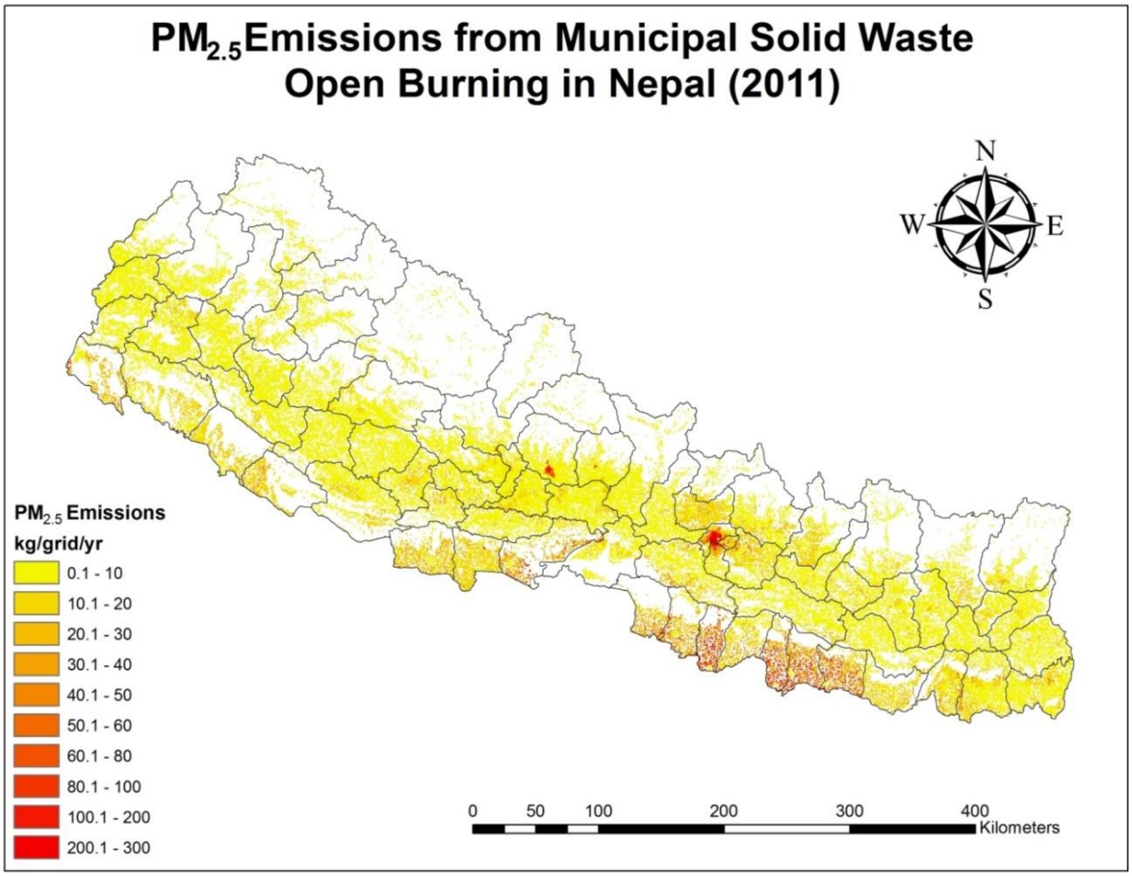

Figure 2.16: $PM_{2.5}$ emission flux from MSW open burning for 2011, Nepal

CHAPTER 3

FUEL-BASED EMISSION INVENTORY FROM ROAD VEHICLES IN THE KATHMANDU VALLEY AND NEPAL

Abstract

This study estimates emissions from diesel vehicles from 1989-2018 referring data from national statistics, average vehicle kilometers traveled, fuel mileage, and experimental-based emission factors (EFs) for each vehicle category at idling and on-road settings. Ratnoze1 and Microaeth AE51 were used to measure the tail pipe emissions. Total diesel usage in 2017/18 in Nepal was estimated to be 892,770 kL (85-115%), of which 29.3% is for the Kathmandu Valley. The carbon mass balance technique was used to compute the fuel-based EFs of CO_2, CO, BC, and $PM_{2.5}$. Of total vehicles studied (n = 29) in idle condition, the average EFs were calculated as CO_2 2600 (99-101%), CO 33.3 (44-156%), BC 0.6 (25-101%), and $PM_{2.5}$ 5.2 (0-235%) in unit of g L^{-1}. For moving conditions (n = 5), the average EFs were estimated as 2,476 for CO_2 (90-110%), 97.3 for CO (0-232%), 1.7 for BC (46-110%), and 20.7 for $PM_{2.5}$ (0-255%), all in g L^{-1}. The estimated national air pollutant emissions were calculated by multiplying diesel consumption by EFs, which were CO_2 2,214 (90-110%) to 2,781(85-115%), CO 27.7 (42-158%) to 88.8 (0-232%), BC 0.51 (23-177%) to 3.55 (46-110%), and $PM_{2.5}$ 3.42 (0-236%) to 23.47 (0-255%) in the year 2017/18, all in Gg, of which 24.4-29.5%, 28.9-32.3%, 12.3-31.9%, and 21.8-42.5% is respectively for the Kathmandu Valley. The daily temporal emissions profile of 2017/18 indicates CO_2 138.3-147.2 tons, and $PM_{2.5}$ 241.3-1449.2 kg were highest at 2-3 p.m.; however, BC 36.8-178.3 kg, and CO 2.1-5.8 tons had the highest at 11 a.m.-12 p.m. in the valley. This study suggests updating national vehicle mass emission standards and amending National Transportation Policy 2001 to include and enhance sustainable low-carbon transport pathways.

Keywords: Diesel vehicles; Air quality; Emission inventory; Emission factors

3.1 Introduction

The number of people living in urban areas globally in between 1950 and 2011 increased by around five times (i.e., 0.75 billion-3.6 billion) (UN, 2012). Similarly, the use of motor-powered vehicles is increasing (Aggarwal & Jain, 2016; Badami, 2004; Badami, 2005). By 2040, there will be nine billion people on the planet, which will require a 30% increase in energy use from 2016 (ExxonMobil, 2013). In Nepal, the transport sector is expanding the fastest in line with urban growth and socioeconomic development. This sector is one of the major contributors to air pollution, which has been found to significantly worsen air quality (Guo *et al.*, 2016). The increase in air pollution brought on by the transportation industry is a result of urbanization, socioeconomic growth, and population growth (Guo *et al.*, 2016; Wu *et al.*, 2017). Transport sector releases harmful pollutants like $PM_{2.5}$, HC, NO_x, and SO_2 as well as greenhouse gases such as CO_2, N_2O, CH_4 (Garg *et al.*, 2006; Kathuria, 2004; Neeft *et al.*, 1996; Raux, 2004; Ribeiro *et al.*, 2007; Wu *et al.*, 2017). In 2004, the transport sector added 23% of global CO_2 emissions as well as 74% of on-road CO_2 emissions, according to Ribeiro *et al.* (2007), which is closer to the estimate (i.e., 26% of CO_2 emissions) of Chapman (2007). Even though the percentage of heavy duty diesel vehicles (HDDV) is reduced, their emissions still have a substantial impact on the issue of air pollution (Kirchstetter, 1999). Road vehicles emissions also contribute to stratospheric ozone depletion, acid deposition, and climate change (Shrestha, 2018).

The studies suggest various techniques to estimate EFs and emissions globally. Singer & Harley (1996) calculated CO emissions from cars and light duty trucks at the South Coast Air Basin by referring to EFs obtained through remote sensing technique having more than 70,000 in-use vehicles. Referring to the Infrared Remote Sensor, motor vehicle emission inventories were carried out in California (Singer & Harley, 1998; Jimenez *et al.*, 2000). Similarly, in California, a study was conducted on the on-road measurement of fine particle as well as nitrogen oxide from light and heavy duty vehicles (Kirchstetter *et al.*, 1999). The other literature reports measurements of EFs of BC and particle number (PN) in August 2006 by recruiting 226 heavy duty (HD) diesel trucks as they passed through a 1 km long tunnel under a California highway. The distributions of EFs of BC and PN from individual HD trucks were skewed, indicating that a sizable portion of the fleet's operating vehicles is responsible for the majority of the pollution (Ban-Weiss *et al.*, 2009). Furthermore, the statistical, temporal, and spatial distributions of the SCAQS aerosol measurements were conducted at more than 40 locations in Southern California to acquire a database of meteorological, air quality, and visibility measurements (Chow *et al.*, 1994). Likewise, a study on the monitoring of road-traffic emissions was conducted using a mobile laser-based photo acoustic system (Marinov & Sigrist, 2003). Following measurements on more than 300 diesel vehicles, another study was carried out utilizing a portable emission monitoring equipment to create a database of EFs of diesel vehicle for China (Shen *et al.*, 2015).

Using portable emission monitoring tools, it was possible to estimate the actual fuel use and CO_2 emissions of urban public buses in Beijing. In total, 75 heavy duty public bus with a range of fuel types (natural gas, conventional diesel, and diesel hybrid) underwent on-road testing in Beijing. This study quantified the effects of various factors, including as road type, typical speed, load mass, and air conditioning on fuel usage (Zhang *et al.*, 2014). Selecting vehicles such as buses, taxis, private cars, and motorbikes, an estimate of traffic related emissions of CO_2 in Iran's capital city was invented (Kakouei *et al.*, 2012). Six sites in China underwent the study on the features and sources of polycyclic aromatic hydrocarbons and fatty acids in $PM_{2.5}$ particles in the dust season in 2004 (Hou *et al.*, 2006). Ambient air samples were taken yearly using passive air samplers in order to research the features and sources of atmospheric polycyclic aromatic hydrocarbons (PAHs) in Shanghai. The field campaign involved the use of gas chromatography-mass spectrometry (Wang *et al.*, 2009). The Paul Scherrer Institute (Switzerland) designed and constructed a mobile pollutant measurement laboratory for "measuring gas phase and aerosol ambient concentrations with high spatial and temporal resolution" for the measurement of on-road ambient concentrations of a large variety of trace gases and aerosol parameters with high time resolution (< 15 s for most instruments), along with geographic and meteorological data (Bukowiecki *et al.*, 2002). The estimation of vehicle fuel consumption and emission rate data for five light duty vehicles and three light duty trucks as a function of the vehicle's instantaneous speed and acceleration levels was also studied using a number of hybrid regression models (Ahn *et al.*, 2002). There is a reported literature on "Trends in multi-pollutant emissions from a technology-linked inventory for India: I. Industry and Transport Sectors", which was studied by Sadavarte & Venkataraman (2014). An estimation of exhaust emissions of on-road vehicle such as CO and HC has been done for more than 1,000,000 vehicles at 20 different locations around the world using the developed remote sensing technology by the University of Denver (Zhang *et al.*, 1995). Recently, another study on the "Evaluation of heavy-duty vehicle emission controls with a decade of California real-world observations" was carried out from 2011 to 2018, referring to fuel-based BC and NO_x EFs. Both BC and NO_x EFs have decreased by 90% and 31%, respectively, over the past decade (Ruehl *et al.*, 2021). A study on "High resolution vehicular exhaust and non-exhaust emission analysis of urban-rural districts of India" was also recently published, highlighting PM re-suspension as the main form of $PM_{2.5}$ emissions from all road types, with rural roads accounting for an even higher proportion (Tomar *et al.*, 2022).

The study on "Analysis of the vehicle fleet in the Kathmandu Valley for estimation of environmental and climate co-benefits of technology intrusions" was conducted by Shrestha *et al.* (2013b). The information that was gathered was processed into the International Vehicle Emission (IVE) model, which then generated EFs for vehicles. However, this study didn’t account for personal cars, trucks, and non-road vehicles. The study on the estimation of PM_{10} concentration in the working location of traffic police personnel was conducted for Pokhara Sub-Metropolitan City (Bashyal *et al.*, 2008). Likewise, the work was carried on the "characteristics and sources of polycyclic aromatic hydrocarbons in atmospheric aerosols in the Kathmandu Valley, Nepal" to determine the concentrations of 15 priority particle bound PAHs (Chen *et al.*, 2015). Another study on emission from vehicles such as CO_2, CO, NO_X, HCs, SO_2, Particulate Matters (PM_{10}, $PM_{2.5}$, BC and OC), and Dioxin/Furans was conducted referring to EFs and Global Warming Potentials (GWPs) of the pollutants (BC, OC, CO_2, and NO_X) for the Kathmandu Valley (Ghimire & Shrestha, 2014). A rapid urban assessment of air quality was carried out for Kathmandu by Pradhan *et al.* (2012). Similarly, the emission inventory for the transport sector was conducted by Sadavarte *et al.* (2019) for the base year 2011. The EFs obtained were completely based on IVE modelling as well as global-based value. According to the research mentioned above, there are only a few fuel-based EFs that can be used for development of emission inventory of both idling and moving conditions. The emissions inventory of gas phase pollutants (i.e., CO_2 and CO) and particulate matter (i.e., BC and $PM_{2.5}$) from idling as well as moving vehicle categories (i.e., bus, mini bus/truck, truck/tipper, pickup, and micro bus) was examined using experimental-based EFs, statistical information of the nation, average vehicle kilometres travelled (VKT), and fuel mileage.

3.2 Objectives

The general objective of this study was to develop an emission inventory of diesel vehicles for the Kathmandu Valley and Nepal through experimental-based EFs.

The specific objectives were:

I. To estimate vehicle-specific diesel consumption trends of Kathmandu Valley and Nepal from a secondary sources of information.
II. To develop fuel based EFs of pollutants from diesel vehicle categories through carbon balance method.
III. To estimate annual trends of pollutants by vehicle category for Kathmandu Valley and Nepal.

3.3 Materials and Methods

3.3.1 Study sites and vehicle categorization

The on-road vehicle survey was conducted from July 5-8, 2019. To study the on-road vehicle fleet, seven sites were selected within the Kathmandu Valley (Table 3.1). Three sites (i.e., Kaushaltar, Jadibuti, and Dhungeadda) were selected for outside the ring road/long route, two sites (i.e., Syuchatar and Chabahil) for the ring road, and the other two sites (i.e., Sorahkhutte and Kupondole) for inside the ring road (Fig. 3.1). The vehicle categories such as buses, mini buses, micro buses, trucks, mini trucks, tippers, mini tippers, tankers, and pick-ups were identified by referring to photographs (Fig. 3.2).

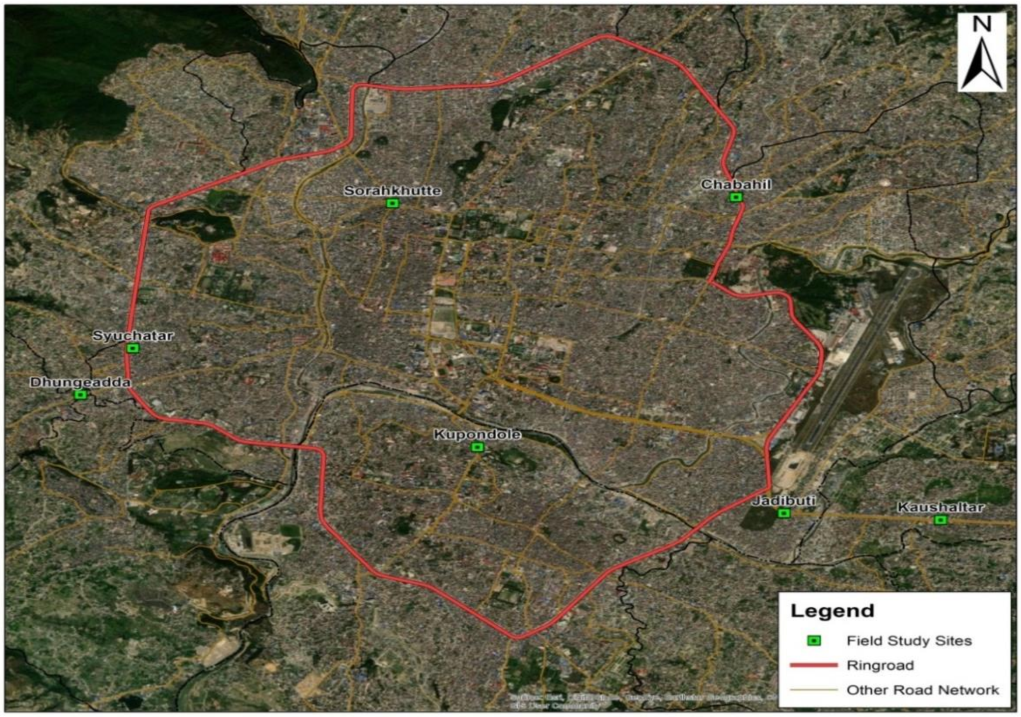

Figure 3.1: Map of field study sites in the Kathmandu Valley (Google Earth)

Then the diurnal profile of diesel vehicle flow variation was estimated for each location every hour. As detailed information about the road vehicle fleet was not available for the year 2017/18, we assumed the same proportionate from the survey year. The year 2017/18 was the most recent data available for the Kathmandu Valley and Nepal. Therefore, it was considered the base year in this study. After the new constituent of Nepal, Department of Transport Management (DoTM) has not updated the statistics.

Table 3.1: Field study sites

Date	Sites	Road	GPS Location	
05/07/2019	Kaushaltar	outside ring road/long route	27°40'27.87"N	85°22'4.36"E
05/07/2019	Jadibuti	outside ring road/long route	27°40'31.20"N	85°21'4.50"E
06/07/2019	Dhungeadda	Outside ring road/long route	27°41'30.98"N	85°16'35.77"E

06/07/ 2019	Syuchatar	Ring road	27°41'54.47"N	85°16'55.73"E
08/07/2019	Chabahil	Ring road	27°43'10.67"N	85°20'46.21"E
08/07/2019	Sorahkhutte	Inside ring road	27°43'7.57"N	85°18'34.97"E
08/07/2019	Kupondole	Inside ring road	27°41'4.64"N	85°19'7.35"E

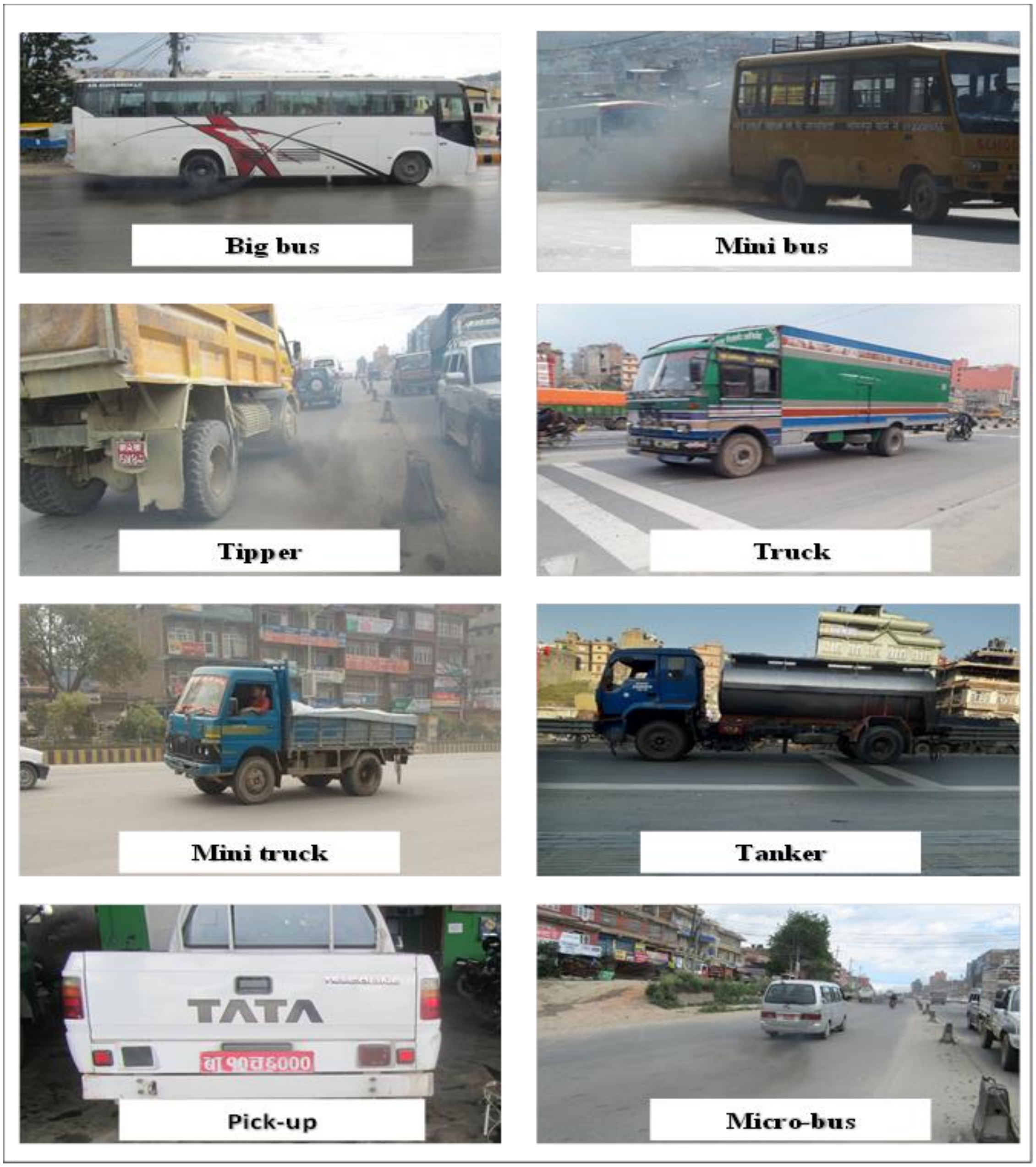

Figure 3.2: Vehicle categorization while vehicle count survey

3.3.2 On-road traffic flow

The number of vehicles plying on the roads of the Kathmandu Valley was estimated at 40 minutes per hour based on a survey conducted every hour between 7 am and 6 pm. Based on field observation, the operational hours of buses, minibuses, pickups, and microbuses were considered to be 1% from 7 pm to 6 am due to scarce public activities outdoors. Similarly, truck and tipper were assumed to be 1% from 10 p.m.-5 a.m., while 7-9 p.m. value was assumed to be the same proportion as the 6-7 p.m. value. The daily pollutants emission (i.e., CO_2, CO, BC, and $PM_{2.5}$) from each vehicle category were distributed as per operational hour of the day.

3.3.3 Vehicle fleet composition

The vehicle census of Bagmati zone/Kathmandu Valley and Nepal from the years 1989/90 up to 2017/18 was obtained from the DoTM (1989-2018) report. The data for the truck/tipper census for Nepal was not available in the literature. However, the accumulated vehicle category (i.e., crane/dozer/excavator/truck/tipper) was reported. From it, we estimated the truck/tipper census of Nepal by referring to the share of construction vehicles from the Bagmati zone and then subtracting the calculated share value from the accumulated vehicle category. The major diesel vehicles considered for the Bagmati zone/Kathmandu Valley were buses, mini-buses, trucks, mini-trucks, pick-ups, and microbuses. Likewise, the major diesel vehicles considered for Nepal were buses, mini buses, trucks, tippers, pickups, and micro buses. The cumulative data of diesel vehicles was estimated from 1989/90 up to 2016/17. In this study, electric trolley buses were excluded from the data. Vehicle survival numbers were calculated by assuming a 20-year vehicle life expectancy based on its performance as well as Nepal's government vehicle ban policy after 2000 (DoTM, 2012). After 20 years, each heading year was subtracted from the initial years in a cumulative manner to get an estimate of the survival number. As the data from 1989/90 is a cumulative value, we redistributed it to the previous six years (i.e., before 1989/90), considering a factor calculated from the average five years from 1990/91 to 1994/95. Moreover, bus and truck/tipper operations were further divided into city-centered and long-route based on the data of a field survey at Dhungeadda/Kalanki and Kaushaltar/Bhaktapur. From 1989/90 to 2017/18, data on diesel vehicle registrations was reordered into Euro grade categories, with an estimated share of Pre-euro (1989-1999), Euro I (1999-2012), and Euro III (> 2012). Because there are no records of Euro II vehicles, they were excluded in this study. The percentage of Euro-grade diesel vehicles is shown in Appendix X.

3.3.4 Vehicle kilometer travelled

Vehicle kilometer travelled was calculated for each selected category (i.e., city center and long route) based on the field survey and available literature (Table 3.2). The VKT of city center vehicles and trucks/tippers on long routes were based on the field study through odometer reading and vehicle registration date. However, the VKT of the long route bus was taken from Bajracharya & Bajracharya (2013).

Table 3.2: Average vehicle kilometer travelled in city center and long route

City center	Avg. VKT/yr.	SD	Min (km)	Max (km)
Bus	23041	15762	5920	48490
Mini bus	23971	22669	7942	40000
Truck/tipper	25178	21311	10109	40247
Mini truck/tipper	21650	12440	7942	40000
Pickup	14088	7511	5105	28124
Micro bus	33223	14899	15267	45608
Long route				
Bus	55617	1710	54407	56826
Truck/tipper	47083	10017	40000	54167

The vehicle mileages were obtained from the published literature (Table 3.3). The average mileage of buses, mini buses, trucks, and mini trucks was obtained from the latest reported value in India (Karali *et al.*, 2019). The average mileage of micro buses and pickups was obtained from the reported value of Nepal (Bajracharya & Bajracharya, 2013; Bajracharya & Bhattrai, 2016).

Table 3.3: Average vehicle mileage

Diesel	Avg. fuel consumption (l/km)
Bus	0.22
Mini bus	0.20
Micro bus	0.16
Truck	0.23
Mini truck	0.20
Pickup	0.15
Truck/Mini truck	0.21

3.3.5 Diesel consumption

Referring to vehicle categories, average VKT and average fuel consumption, as well as annual diesel consumption (i.e., liter/kilo liter) by each vehicle category (i.e., bus, mini bus, truck, mini truck, tipper, mini tipper, pickup, and micro bus) for the Bagmati zone/Kathmandu Valley and Nepal were estimated. The annual consumption of diesel in the transport sector of the Kathmandu Valley and Nepal was compared with the annual sales data from NOC (1993/94-2017/18).

3.3.6 Experimental design and set-up

3.3.6.1 Auto workshops and sample selection

Six auto workshops (Vehicle Fitness Testing Centre, Teku; Department of Transport Management, Ekantakuna; Sajha Yatayat, Pulchowk; Dhungeadda/Kalanki; Sanobharyang/Swayambhu; and Balaju) and three on-roads (Pulchowk-Lagankhel, Pulchowk-Ratnapark, and Sanobharyang-Kalanki) were selected to conduct a study on emission measurement of diesel vehicle from 8 August, 2019 to 30 January, 2020. Overall, 34 vehicles were recruited, of which 11 were buses, 2 minibuses, 2 trucks, 5 mini trucks, 2 tippers, 1 mini tipper, 7 pickups, and 4 micro buses. 29 vehicles were recruited while idling and 5 while moving. In idle condition, 10 were buses, 1 was a mini bus, 2 trucks, 3 mini trucks, 2 tippers, 1 mini tipper, 6 pickups, and 4 micro buses. While in moving condition, 1 was a bus, 1 a mini bus, 1 a pickup, and 2 mini trucks.

3.3.6.2 Instrumentation

Various instruments are reported in the literature that measure air pollutants from the transport sector. Nondispersive infrared sensor (NDIR) was used to measure CO/CO_2 and HC/CO_2 for motor vehicles for on-road study (Zhang *et al.*, 1995). It was also used for the measurement of CO for HDDV through the chassis dynamometer (McCormick *et al.*, 2000; Robert *et al.*, 2007). Likewise, Infra-Red Remote Sensor was used for the measurement of HC for HDDV for on-road study (Singer & Harley, 1998). Furthermore, it was used by Singer & Harley (2000) for measurement of CO, CO_2 and HC for HDDV for on-road study. Chemiluminescence analyzer has been used for measurement of NOx for HDDV for on-road study and through a chassis dynamometer (McCormick *et al.*, 2000; Zhang *et al.*, 1995). A Flame Ionization Detector (FID) was utilized to measure HC for motor vehicles in an on-road study (Zhang *et al.*, 1995). It was also used for the measurement of THC for HDDV through the chassis dynamometer (McCormick *et al.*, 2000; Robert *et al.*, 2007). LANCOM was used for the measurement of NOx and CO for passenger cars through an on-road study (Pujadas *et al.*, 2004). A TSP sampler was used for the measurement of PM for heavy duty diesel vehicles using a Pallflex T60A20 70 mm filter through a chassis dynamometer (McCormick *et al.*, 2000). It was used by Cui *et al.* (2017) to measure PM for heavy duty excavators and trucks through an on-road study. MOUDI was used for the measurement of the mass of PM for HDDV through the chassis dynamometer (ARAI, 2007). The chassis dynamometer was used to measure CO, CO_2, NO_x, THC, and CH_4 for HDDV using the Horiba 9200F, Fisher Rose Mount NGA Analyzer, and Horiba 7200H (ARAI, 2007).

From the above, we understood that CO, CO_2, HC, NO_x, and PM remained the main sources of emissions from diesel vehicle emissions. Moreover, we infer that for the same pollutant measurement, instruments are different as per the authors. In our study, we used Ratnoze 1 to estimate EFs by the carbon mass balance technique. A sample train is a set of entrapment devices, instruments, and auxiliary apparatus that are arranged in a specific order to selectively separate and collect samples of specific air pollutants. Two parallel 47-mm filter holders and a selective intake that was sized for $PM_{2.5}$ were included in the sample train, which was used to gather filter samples for laboratory analysis. Several sensors were installed on Ratnoze 1 to measure different pollutants, including CO (using an electrochemical sensor), CO_2 (using an NDIR sensor), and PM (through light absorption, MicroAeth). To validate our results, we used E-Instruments Model 8500 Plus, which was fitted with different sensors for the measurement of pollutants. Moreover, we used LI-820 for CO_2 measurement, fitted with an NDIR sensor (Fig. 3.3).

3.3.6.2.1 Ratnoze 1

Ratnoze 1, manufactured by Mountain Air Engineering, is a portable instrument that is used for measurement of emissions from fuel combustion. EFs are calculated using the carbon mass balance approach. Real-time measurements of gaseous and particle pollutants are done using sensors. It collects particulate samples in its two filter holders for further lab analysis. Ratnoze not only measures pollutants from exhaust plume gas but also measures the background gas. Data is logged at a one-second time base in the internal memory, which can be viewed and plotted during the sampling duration for visualization of data. The data can be easily downloaded and plotted as required (Mountain Air Engineering, 2019).

3.3.6.2.2 Micro-Aeth AE51

The Micro-Aeth AE51, manufactured by AethLabs, is a highly sensitive portable instrument that is used to measure the optically absorbing BC component of aerosol particles. Using a light absorption technique based on filters, it analyzes black carbon in real time. The non-volatile memory stores the BC mass concentration at the time of the operation. The AE51 pump operated at a 50 ml/min sampling flow rate setting with data logged every ten seconds is appropriate for traffic and transportation impacts at higher BC concentrations. Data can be downloaded after sampling using the USB interface cable (AethLabs, 2016).

3.3.6.2.3 E instruments (E-8500 Plus)

E instruments (E-8500 Plus) consist of electrochemical sensors: nitrogen dioxide sensor (NO_2), nitric oxide sensor (NO sensor), sulfur dioxide sensor (SO_2); hydrogen sulfide sensor (H_2S), oxygen sensor (O_2 sensor), and carbon monoxide (CO) sensor. Additionally, it has PID (Photo-Ionization) sensors to detect VOCs as well as NDIR (infrared) sensors to monitor CO of high range, VOC and CO_2.

3.3.6.2.4 LICOR (LI-820)

The LI-820 CO_2 analyzer was created by LICOR for applications requiring continuous monitoring. The LI-820 is a non-dispersive infrared (NDIR) gas analyzer solely based on a single-path, dual wavelength infrared detection system.

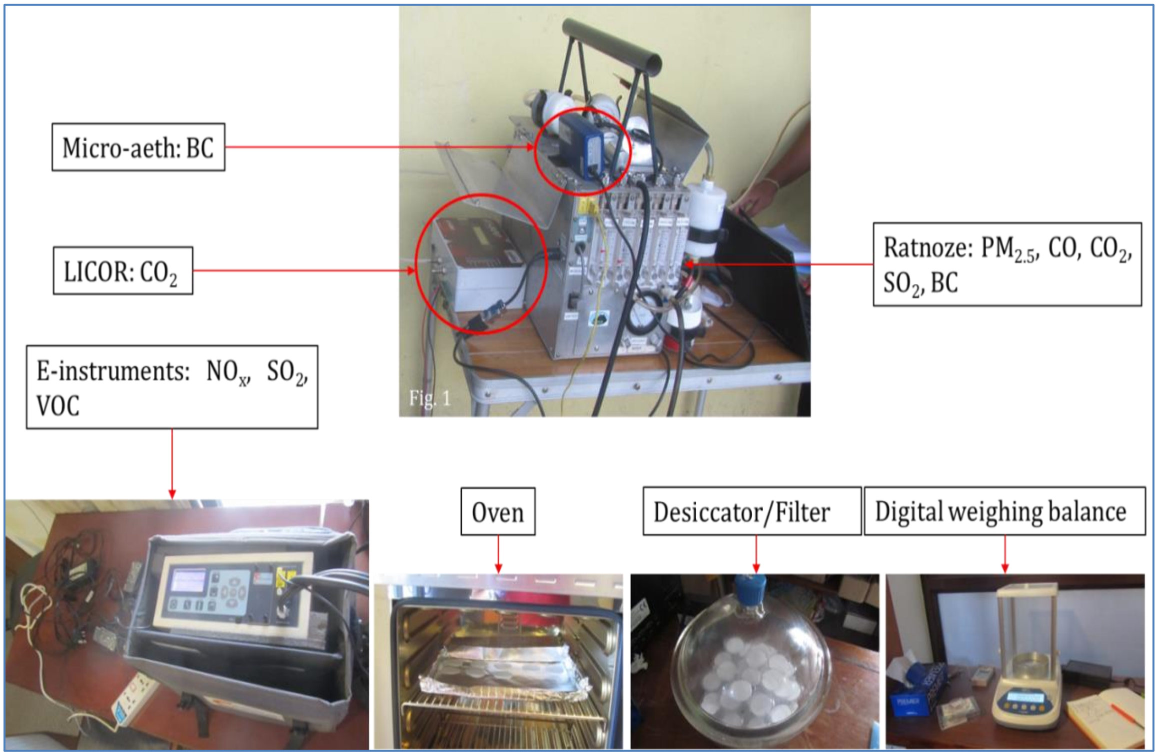

Figure 3.3: Emissions measurement equipment

3.3.6.3 Sampling technique and laboratory analysis

3.3.6.3.1 Field preparation

i. The batteries of sensor box, Micro-Aeth and laptops (both Ratnoze and MicroAeth) were charged prior to the field measurement.
ii. The Quartz filter paper and Polytetrafluoroethylene (PTFE) were both ready for the sampling study. Approximately 8 hours of desiccation in a desiccator were followed by keeping quartz filters in a hot air oven set at 310^0C. PTFE filters were only desiccated for 24-hours to remove moisture from it.
iii. The filter papers were first weighed using a digital balance before being put in a sterile petri dish, correctly labeled, and taped shut with Teflon tape.

3.3.6.3.2 Calibration of instrument

The gaseous and particulate sensors of Ratnoze 1 were factory calibrated. Before sampling, the instrument was validated through calibration gas mixtures. The sole onsite zero calibration that was done prior to tailpipe emission measurements was through the high efficiency particulate air (HEPA) filter. BC was measured using a Micro-Aeth that was factory calibrated. Specialty Gases Ltd., West Bromwich, UK, provided a zero air gas for the CO gas sensors to be verified at 0 ppm, and Alchemic Gases and Chemical Pvt. Ltd., Mumbai, India provided a CO calibration gas combination for the sensors to be validated at 81 ppm. The CO_2 sensor was also verified using a CO_2 calibration gas mixture at 350 PPM and 1250 PPM (Specialty Gases Ltd.). CO_2 data were collocated with LICOR (LI-820, Lincoln, Nebraska, USA) during pre and post field experiments for validating.

3.3.6.3.3 Pre-sampling setup

i. The probe was assembled.
ii. Filter papers were loaded in their respective filter holder.

iii. Leak test was performed on main inlet and dilution inlet using vacuum gauge.

iv. To keep background PM from gathering on the sample filters, the HEPA filter was connected to the cyclone inlet.

v. The serial data output was allied to a laptop running Ratnoze software using a USB cord.

vi. The sensor box was powered on.

vii. After a new Micro-Aeth filter was loaded on the Mico-Aeth, its valve on the Ratnoze was opened and finally Micro-Aeth was switched on.

viii. The software was run while Micro-Aeth was linked to its laptop.

ix. The clocks were synced for Ratnoze, Micro-Aeth and laptops.

x. To warm up, the sensor box was run for ten minutes.

xi. The calibration and zeroing of all the flow sensors (F1, F2, Gas flow, and Dilution flow) was done.

xii. It was decided to enter the barometric pressure displayed by the Kestrel meter as the absolute stack pressure in the Ratnoze calibration parameters.

xiii. The flows were set as
F1 flow = 400 sccm
F2 flow = 400 sccm

Dilution flow adjusted according to the desired dilution ratio which varied on different conditions. The gas flow was adjusted in such a way that the cyclone flow was 1500 sccm (Adhikari *et al.*, 2019; Mool *et al.*, 2020).

3.3.6.3.4 Exhaust sample period

Emissions of various categories of diesel vehicles were studied with the help of the portable sampling system named Ratnoze1 (Mountain Air Engineering, 2016) (Appendix XXIII). This instrument was used in local maintenance centers to measure both particulate and gaseous pollutants from diesel vehicle tailpipes. The device was used to measure background CO_2 and CO concentrations as well as the real-time concentrations of CO, CO_2, BC, and $PM_{2.5.}$ The importance of the use of background concentrations is reported in Mool *et al.* (2020). The sample train consists of two parallel 47-mm filter holders that were used to collect filter samples for additional laboratory analysis, as well as a selective inlet that is sized for $PM_{2.5}$. To collect the aerosol particles inside the filter holders for gravimetric analysis, two filter papers—Pall Corporation Teflo filter membranes and quartz fiber filter membranes (of 47 mm in diameter with a 2 m pore size)—were utilized. The more detail description on the dilution system, flow movement and calibration of all the sensors used in the instrument are found in Adhikari *et al.* (2019). Depending on a filter load, samples of the diesel vehicles' emissions were taken for more than 15 minutes while they were idling and for about 5 minutes while they were moving.

i. Pre-background reading was taken for at least 10 minutes.
ii. The probe was moved so that the nozzle was right in front of the tailpipe opening and the tubes were connected immediately after removing the HEPA filter on the cyclone inlet. The sampling start time was noted.
iii. During the sampling period, all the sensor flows and the Micro-Aeth ATN was watched and adjusted if needed. Micro-Aeth filter was changed when ATN value exceeded 120.
iv. During the end of sampling time, probe was disconnected from the sensor box and was moved away from the tailpipe. The end time was also noted.
v. HEPA filter was connected to the cyclone inlet.
vi. The sensor box was left to operate for 10 more minutes to take post-background sampling.
vii. The data from Micro-Aeth was downloaded to a folder and then powered off.
viii. Ratnoze data was also downloaded and the sensor box was powered off.

3.3.6.3.5 Post-sampling tasks

i. Both filter papers were taken out of their filter holders, put in the same petri dish they had come from, properly sealed with Teflon tape, and put in a zip-lock bag.

ii. All the equipment and samples were transported to the assigned laboratory for further analysis.

iii. In the laboratory, probe, cyclone and sensor box were cleaned and batteries were charged.

iv. After being desiccated for 24 hours, the filters were weighed using a digital balance.

3.3.7 Data analysis

The average fuel-based EFs of the diesel vehicle categories were calculated through the carbon mass balance technique. The carbon mass balance method's basic principles and detailed EF calculations are described in the reported studies (Adhikari *et al.*, 2019; Mool *et al.*, 2020). To calculate fuel-based EFs for pollutants, the carbon mass balance method was considered (Stockwell *et al.*, 2016). According to this technique, the EFs are normalized to fuel utilization (i.e., grams of pollutants emitted per liter of fuel consumed) (Singer & Harley, 1996; Stockwell et al., 2016). If the molar exhaust concentrations of CO, CO_2, and BC are measured, it is possible to use the carbon balance method to connect the amount of pollutants emitted to the quantity of fuel used (Singer & Harley, 1996; Stockwell *et al.*, 2016). By carbon balance, it is possible to relate the amount of pollutants emitted to the amount of fuel burned if the molar exhaust concentrations of CO_2, CO, and BC are measured (Singer & Harley, 1996; Liu *et al.*, 2009).

$$\mathrm{EF_P} = \left(\frac{[\mathrm{P}]}{\mathrm{Total\ Carbon}}\right) * \frac{\mathrm{w_c}\rho_\mathrm{f}\mathrm{M_P}}{12} \qquad (3.1)$$

Where, EF_P represents EF of pollutant P (in units of grams of pollutants per unit volume of fuel consumed), [P] represents exhaust concentration of pollutant P, w_c represents carbon weight fraction of the fuel, ρ_f represents fuel density, and M_P represents molecular weight of P. The factor of 12 is the carbon atomic mass.

EF calculation (CO_2 and CO) is as below:

$$\mathrm{EF}_{\mathrm{CO_2}}(\mathrm{g/L}) = \left(\frac{\text{Exhaust concentration}(\mathrm{C}_{\mathrm{CO_2}})}{\text{Total Carbon}}\right) * \frac{\mathrm{w_c} * \rho_\mathrm{f} * 44}{12} * 1000 \quad (3.2)$$

$$\mathrm{EF}_{\mathrm{CO}}(\mathrm{g/L}) = \left(\frac{\text{Exhaust concentration}(\mathrm{C}_{\mathrm{CO}})}{\text{Total Carbon}}\right) * \frac{\mathrm{w_c} * \rho_\mathrm{f} * 28}{12} * 1000 \quad (3.3)$$

Where, w_c is 87%, and ρ_f is 0.832 kg L^{-1}. Since Nepal's w_c value was not provided, we used the USA-based value instead (Kirschtetter *et al.*, 1999), which was too referred by Adhikari *et al.* (2019) and Mool *et al.* (2020) for Nepal. Likewise, value of ρ_f (in average) for the BS III standard was taken from India (CPCB, 2010).

The EF of particulate pollutants (BC and $PM_{2.5}$) was calculated with reference to CO EF. EF_{CO} was converted to EF for fine particle mass (EF_{PM}) by the ratio of filtered PM mass and the corresponding mass of CO obtained from the filters (Jayarathne *et al.*, 2017).

$$\mathrm{EF}_{\mathrm{PM}} = \frac{\mathrm{PM}(\mu\mathrm{g/m^3})}{\text{Emitted concentration }(\mathrm{C}_{\mathrm{CO}})} * \mathrm{EF}_{\mathrm{CO}} \quad (3.4)$$

$$\mathrm{EF}_{\mathrm{BC}} = \frac{\mathrm{BC}(\mu\mathrm{g/m^3})}{\text{Emitted concentration }(\mathrm{C}_{\mathrm{CO}})} * \mathrm{EF}_{\mathrm{CO}} \quad (3.5)$$

Total carbon was calculated as follow:

$$\text{Total carbon }(\mu\mathrm{g/m^3}) = \mathrm{C}_{\mathrm{CO_2}} + \mathrm{C}_{\mathrm{CO}} + \mathrm{BC} \quad (3.6)$$

Where, C_{CO_2} is the concentration of carbon in CO_2 ($\mu g/m^3$); C_{CO} is the concentration of carbon in CO ($\mu g/m^3$); and BC is the concentration of carbon in BC ($\mu g/m^3$).

The CO_2 and CO concentrations was measured in ppm, therefore, it was converted into $\mu g/m^3$ through ideal gas equation, $PV = \left(\frac{m}{M}\right) RT$

Exhaust concentration of carbon in CO_2 and CO were estimated as below:

$$C_{CO_2}(\mu g/m^3) = \frac{CO_2(ppm)*Pressure\ (Pa)*12}{R\ (JK^{-1}mol^{-1})*Temp\ (K)} \quad (3.7)$$

$$C_{CO}(\mu g/m^3) = \frac{CO(ppm)*Pressure\ (Pa)*12}{R\ (JK^{-1}mol^{-1})*Temp\ (K)} \quad (3.8)$$

Where, R is universal gas constant (i.e., 8.314 $JK^{-1}mol^{-1}$) and Temp (K) is the temperature of Nozzle.

A series of data from Ratnoze 1 that were logged at a base rate of one second were averaged out using the MS Excel function, and converted into one minute on average. It was also used to analyze data from the Micro-aeth, LICOR, and E-instruments in the same line. The data was incisively analyzed and the results were presented through charts and graphs using MS Excel. The Monte Carlo technique was used to calculate the range of uncertainties of diesel utilization as well as air pollutants from diesel vehicle categories of Nepal (Appendix XI).

It was not possible to conduct emission measurements of pollutants for trucks/tippers and microbuses in moving condition. Therefore, it was estimated by referring to similar-category vehicles as well as available literature. To estimate the emissions for moving vehicles, a factor was required from other study. The lower reported value of truck/tipper CO_2 as well as the higher value of CO were estimated referring to the mini truck/tipper; however, BC and $PM_{2.5}$ (higher value) were estimated based on the idling situation of truck/tipper from the experiments (Nepal-based) and the idling and on-road situation from USA (Park *et al.*, 2011). Likewise, it encountered a problem while measuring $PM_{2.5}$ in moving conditions. Therefore, referring to $PM_{2.5}$ and BC at idling and BC at moving situation, the $PM_{2.5}$ EF (on-road circumstance) was estimated. Additionally, emission measurement of pollutants from micro buses in moving conditions was not possible to conduct. In this case, we estimated EFs for moving conditions by referring to EFs of pollutants of mini buses (idling condition), micro buses (idling condition), and mini buses (moving condition).

From the overall estimated EFs, they were further divided into Euro I (~Bharat Standard, BS I) and Euro III (BS III) diesel vehicles. Because pre-euro grade vehicles were not available throughout the study investigation, scaling factor (SF) was used for the calculation of EFs to estimate the yearly trend of emissions. Referring to the published regional and global-based EFs, SF by euro-grade vehicle was determined, as shown in Appendix XVI. The CO_2 EF and CO EF for trucks, buses, and light commercial vehicles (LCV) for the pre-euro circumstance are available in Indian context (ARAI, 2007). The CO_2 EF for LCV and buses (Euro I) was taken from ARAI (2007) and Gurjar *et al.* (2004), whereas for trucks, it was based on ARAI (2007). For buses (Euro III), CO_2 EF was obtained from the literature (Yang *et al.*, 2016), whilst LCV was computed proportionately from a bus, and a truck stood same as buses. The CO EF for Euro I bus was obtained from the available studies (ARAI, 2007; Mashelkar *et al.,* 2002; Bajracharya & Bhattarai, 2016). The CO EF for trucks (Euro I) was taken from ARAI (2007) and for Euro I and III trucks it was considered from Mahesh *et al.* (2019). CO EF for LCV for Euro I was taken from the reported studies (ARAI, 2007; Bajracharya & Bhattarai, 2016; Mashelkar et al., 2002). The CO EF for LCV (Euro III) was calculated by considering to proportional data from the trucks.

Pre-euro BC EFs for buses were calculated proportionally to trucks (Kirchstetter *et al.*, 1999). Likewise, EF of BC of diesel LCV (for Pre-euro) was estimated referring to the average value of trucks and buses. The reported value of HDDV by Allen *et al.* (2001) was considered to estimate BC EF for buses (Euro I) and Euro III (Raparthi & Phuleria, 2021). The EF of BC for Euro I trucks was taken from the literatures (Geller *et al.*, 2005; Ban-Weiss *et al.*, 2008), while the EF of BC for LCV and Euro III trucks were estimated by referring proportionate value from buses (Allen *et al.*, 2001; Raparthi & Phuleria, 2021). The BC EF for LCV (Euro I) was calculated referring the average value of buses as well as trucks. The $PM_{2.5}$ EF of buses and LCVs (Pre-euro and Euro I) was provided by the literatures (ARAI, 2007; Pandey & Venkataraman, 2014); however, $PM_{2.5}$ EF of trucks was considered to be the same as buses. Furthermore, the $PM_{2.5}$ EF of Euro III vehicles was calculated using a linear regression model. Based on various modes of transportation, the following equation was used to compute the emissions from diesel vehicles (Pandey and Venkataraman, 2014).

$$\mathrm{E_m} = \sum\nolimits_{\mathrm{f,t}} \mathrm{F_{m,f,t}} \times \mathrm{EF_{m,f,t}} \quad (3.9)$$

Where, E_m is the total emissions (in Gg/y) from mode m, and $F_{m,f,t}$ and $EF_{m,f,t}$ are the fuel consumption (in MT/y) and emission factor (in g/kg fuel burned) respectively, for fuel type f and combustion technology t within that mode.

3.4 Results and Discussion

3.4.1 Traffic condition of Bagmati zone/Kathmandu Valley and Nepal

The year 1989/90 was the first officially registered vehicle in Nepal by DoTM. Therefore, this year was taken into account in this study. From 1989/90 until 2017/18 (the most recent data), a total of 3,221,042 vehicles have been registered in Nepal. Of them, total registration in the Bagmati zone/Kathmandu Valley was 1,172,413 (i.e., 36.4%) (DoTM, 1989-2018). Another study in the past showed motorcycles (MCs) having a highest share of Euro III grade (i.e., 75%) and the remaining grades were Euro II and a lower. MCs accounted for the largest share of the operating fleet (56-90%), followed by buses (4-17%), vans and taxis (5-14% each), and 3-wheelers (1.6-4.5%) in the valley (Shrestha *et al.*, 2013b).

The number of registered diesel vehicles (buses, minibuses, trucks, mini trucks, pickups, and microbuses) in the Bagmati zone/Kathmandu Valley was 75,436 in 2017/18, a 22.5 times higher with respect to 1989/90 (DoTM, 1989-2018) (Fig. 3.4). Pickups experienced the highest growth (66.6 folds) between 2002/03 and 2017/18, followed by micro buses (22 folds) during the same period; and buses (16.3 folds), trucks (14.3 folds), mini trucks (14.3 folds), and mini buses (13 folds) from 1989/90 to 2017/18. Similarly, total registered diesel vehicles in Nepal (i.e., buses, mini bus/truck, truck/tipper, pickup, micro bus) was 270,723, an increase of 16.4 folds with respect to 1989/90 (DoTM, 1989-2018)

(Fig. 3.5).The pickup had the most growth (96.3 folds) between 2002/03 and 2017/18, followed by the micro bus (33 folds) during the same period; and truck/tipper (12.7 folds), mini bus/truck (12.4 folds), and bus (12.5 folds) between 1989/90 and 2017/18.

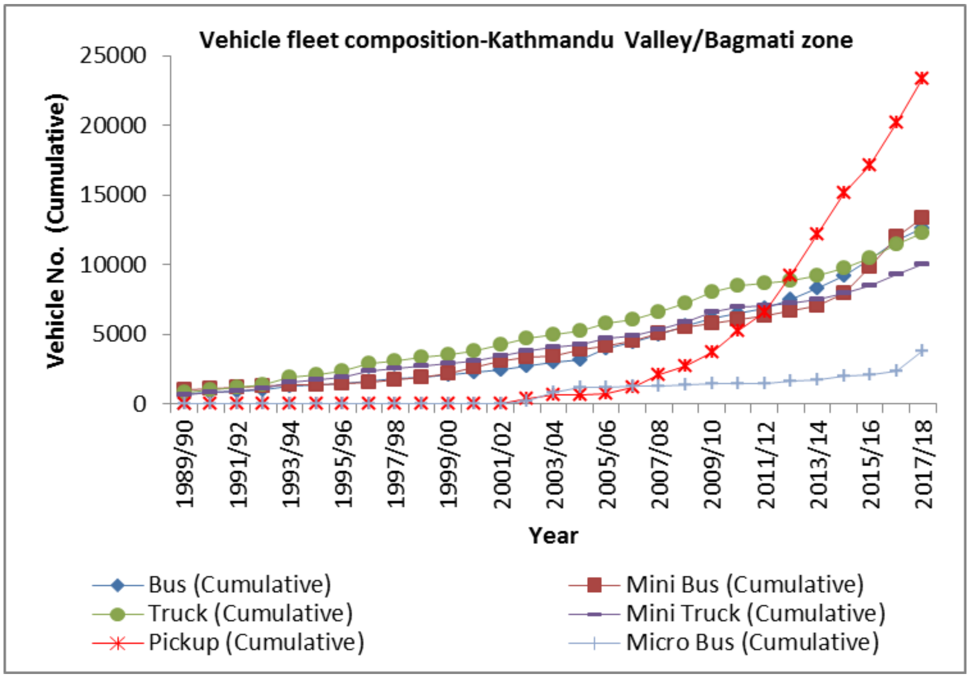

Figure 3.4: Vehicle fleet composition in Bagmati zone/Kathmandu Valley (DoTM, 1989-2018)

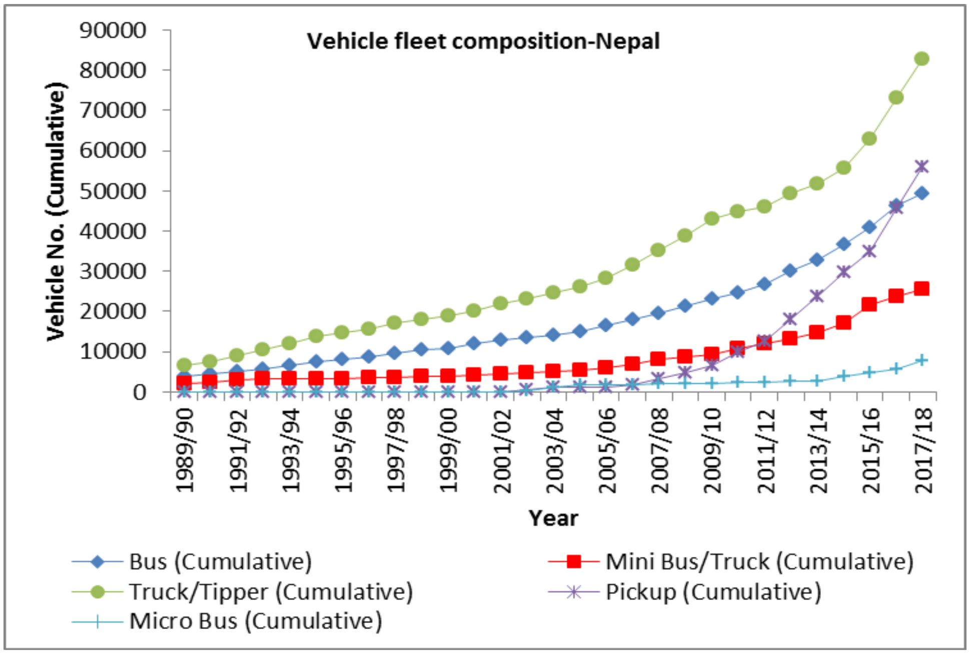

Figure 3.5: Vehicle fleet composition in Nepal (DoTM, 1989-2018)

3.4.2 Vehicle fleet composition

In the Kathmandu Valley, 23,575 diesel vehicles were physically counted in seven distinct locations. Mini buses accounted for the greatest proportion of total observed vehicles, followed by pickups, micro buses, buses, trucks, truck/tankers, mini trucks/tankers, tippers, construction vehicles, and mini-tippers (Fig. 3.6).Of total diesel vehicles, 7,607 vehicles were found to be high emitters, which was 32% of the total counted vehicles. Among all the emitters, the mini bus occupied the highest proportion, followed by the micro bus, pickup, bus, truck/tanker, mini truck/tanker, tipper, construction vehicles, and mini tipper (Fig. 3.7).

The highest hourly average number of vehicles flow in a day (7 am to 6 pm) was for mini buses, at 544/hr (high emitter 238/hr), followed by pick-ups (total 478/hr and high emitter 98/hr), micro buses (total 360/hr and high emitter 104/hr), buses (total 255/hr and high emitter 84/hr), truck/tanker (total 150/hr and high emitter 58/hr), mini truck/tanker (total 140/hr and high emitter 38/hr), tippers (total 25/hr and high emitter 9/hr), construction vehicles (total 9/hr and high emitter 3/hr), and mini tipper (total 4/hr and high emitter 2/hr).

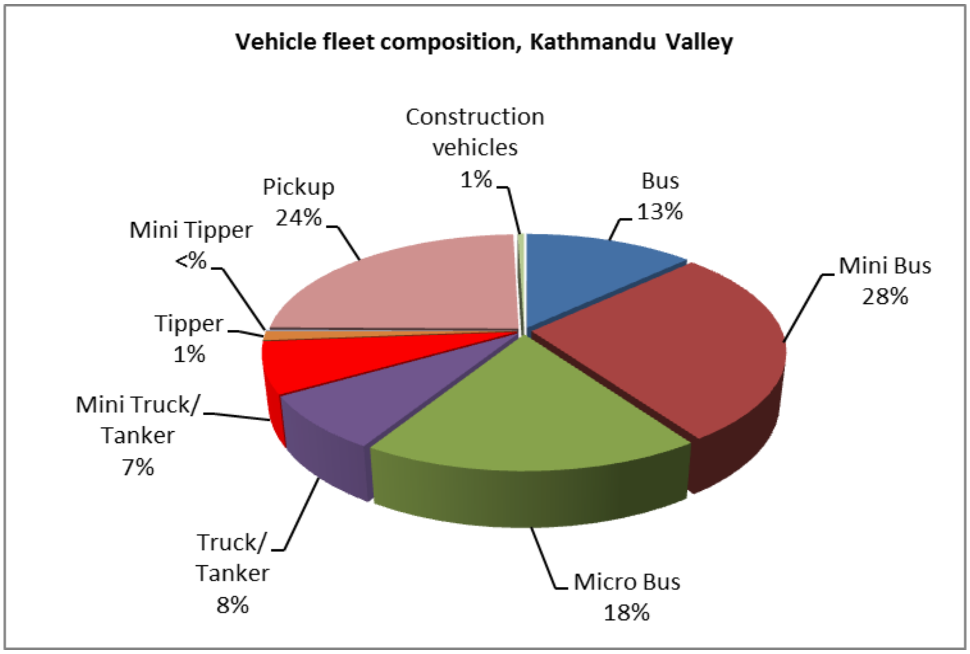

Figure 3.6: Vehicle fleet composition of Kathmandu Valley

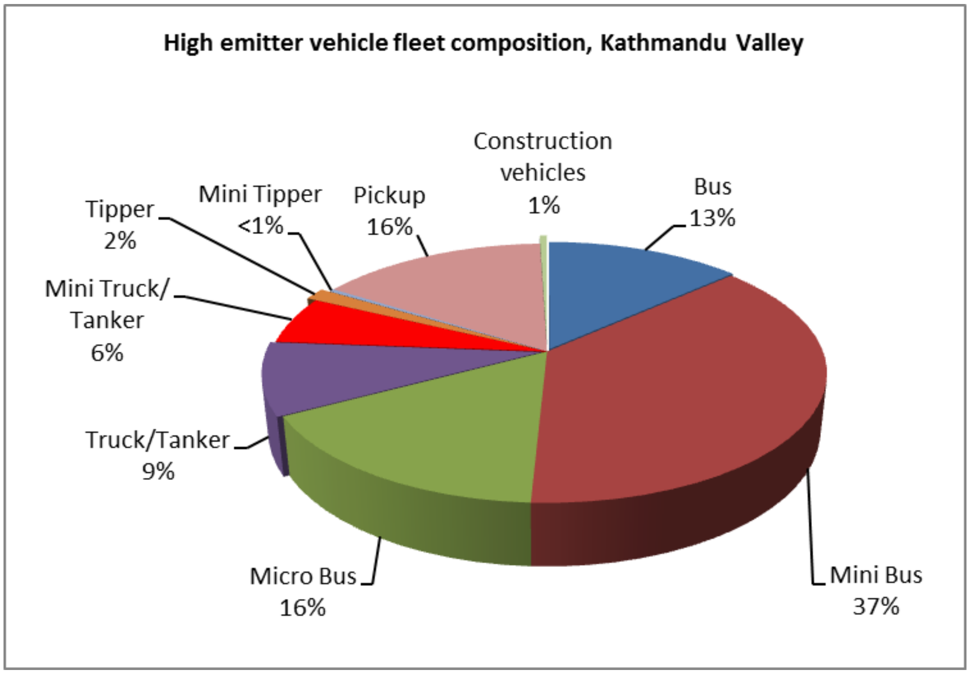

Figure 3.7: High emitter vehicle fleet composition of Kathmandu Valley

3.4.3 Diesel consumption

On average, the trends of diesel consumption by vehicle category in the Bagmati zone/Kathmandu Valley have increased from 1989/90 to 2017/18 (Fig. 3.8). It was estimated to be 261,525 kL in 2017/18, 17 times higher than the value for 1989/90. Of the total diesel sold by Nepal Oil Corporation (NOC) in 2017/18, 16.4% was used in diesel vehicles in the Bagmati zone/Kathmandu Valley. Likewise, on average, the trends of diesel consumption in Nepal have increased from 1989/90 to 2017/18 (Fig. 3.8). It was estimated to be 892,766 kL in 2017/18, 13.4 times higher than the value for 1989/90. Of the total diesel sold by NOC in 2017/18, 55.9% was used in diesel vehicles in Nepal (Das *et al.*, 2022). In the years 2007/08, diesel sales were comparatively low despite increased diesel vehicle registration growth for the same year and the previous year. Likewise, in the year 2015/16, sales of diesel declined considerably. It was due to an earthquake disaster as well as disruptions in petroleum supply caused by political unrest on the Nepal-India border (Underwood *et al.*, 2020). The estimated diesel consumption has been compared with the findings from the available literature. The average diesel consumption by the transport sector was 60% from the years 1993/94-2017/18, which is in line with the published literature. According to the World Bank (2014), 30-40% of all diesel used to generate electricity in Nepal during the electricity crisis. As of WECS (2010), in the year 2008/09, diesel consumption by the transport sector was 357,000 kL (80%), which is close to this study's (i.e., 323,000 kL; 72.4%). The diesel consumption value for the year 2011/12 as reported by WECS (2014) (409,000 kL; 65%) is also in line with this study; 460,000 kL (71%) (Fig. 3.8). Furthermore, it was compared with Sadavarte *et al.* (2019). The findings showed that the values for 2001/02 (> 6.6%), 2005/06 (> 4.2%), and 2016/17 (< 3%) were all very close. However, the value of 2011/12 was higher by 25.4% and 12.5% with respect to Sadavarte *et al.* (2019) and WECS (2014), respectively. It is highlighted that the excluding of other diesel-powered vehicles from this study, such as ambulances and construction vehicles may have had a little impact on the findings.

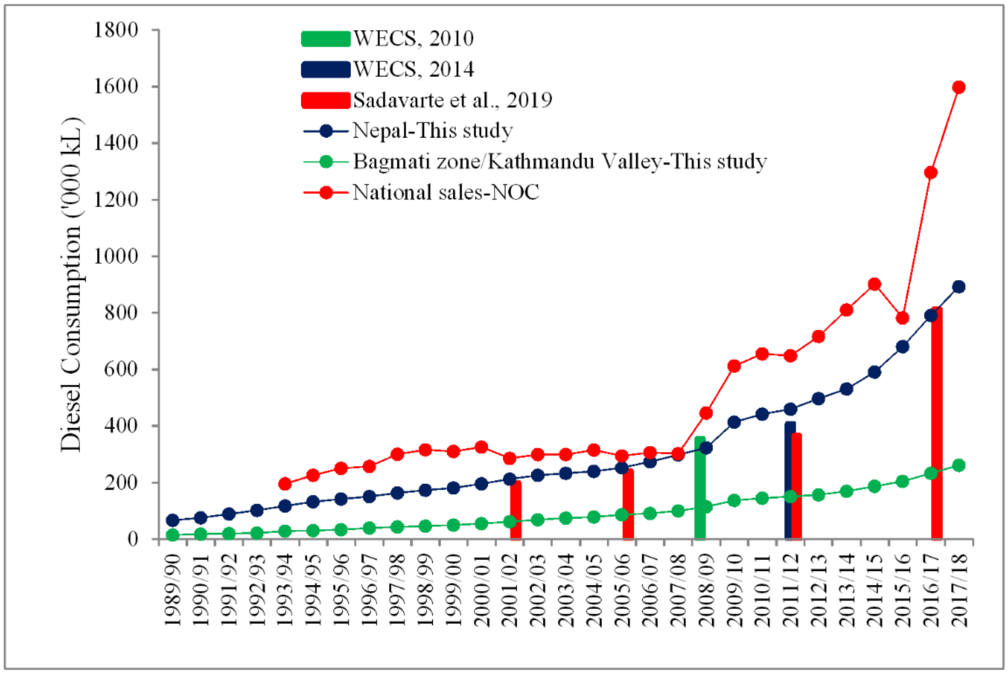

Figure 3.8: Status of diesel consumption in Bagmati zone/Kathmandu Valley and Nepal

3.4.3.1 Diesel consumption by vehicle category in the Kathmandu Valley

On average, the trends of diesel consumption by buses in the Bagmati zone and Kathmandu Valley have increased from 1989/90 to 2017/18 (Appendix XII). It was estimated to be 58,010 kL in 2017/18, 14.8 times higher than the value for 1989/90.

Similarly, diesel consumption by mini bus in 2017/18 was estimated at 47,548 kL, truck/tipper (55,740 kL), and mini truck/tipper (30,621 kL), representing increases of 12.1, 11.5, and 11.5 folds, respectively, compared to 1989/90. Diesel consumption by pickup (49,395 kL) and micro bus (20,210 kL) increased by 66.6 and 22 folds, respectively, in 2017/18 compared to 2002/03, the first year registration began (Fig. 3.9) (Das *et al.*, 2022).

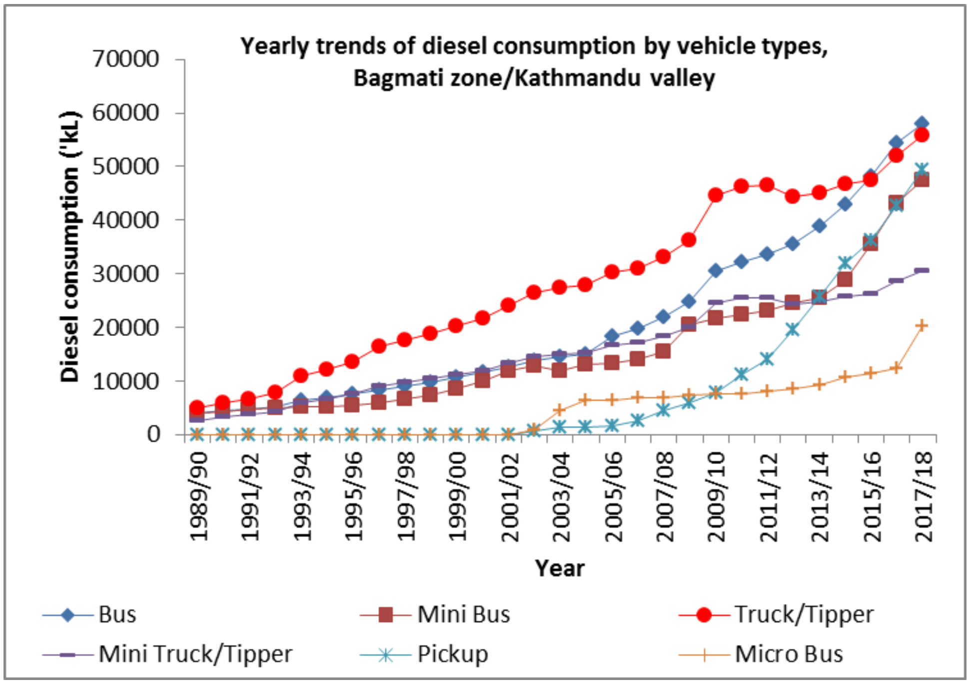

Figure 3.9: Status of diesel consumption in Bagmati zone/Kathmandu Valley by vehicle category

Likewise, in the year 1989/90 in the Kathmandu Valley, the highest share of diesel consumption was by truck/tipper (i.e., 31.6%), followed by mini bus (i.e., 25.6%), bus (i.e., 25.5%), and mini truck/tipper (i.e., 17.4%). Since pickups and microbuses were not deployed in the Kathmandu Valley during this year, the diesel consumption was estimated at zero. However, in the year 2017/18, buses represented the highest share of diesel consumption (i.e., 22.2%), followed by trucks/tippers (i.e., 21.3%), pickups (i.e., 18.9%), mini buses (i.e., 18.2%), mini truck/tippers (i.e., 11.7%), and micro buses (i.e., 7.7%) (Das *et al.*, 2022). The diesel consumption of pickups exceeded that of microbuses and mini trucks/tippers due to the increased number of registrations (Fig. 3.9).

3.4.3.2 Diesel consumption by vehicle category in Nepal

On average, the trends of diesel consumption by buses in Nepal have increased from 1989/90 to 2017/18 (Appendix XIII). It was estimated to be 223,606 kL in 2017/18, 10.8 folds higher than the value for 1989/90. Similarly, diesel consumption by minibus/truck in 2017/18 was estimated at 98,068 kL and truck/tipper (412,100 kL), representing an increase of 11.6 and 10.9 folds, respectively, compared to 1989/90. Diesel consumption by pickup (118,284 kL) and micro bus (40,708 kL) increased by 96.3 and 33 folds, respectively, in 2017/18 compared to 2002/03, the first year registration began (Fig. 3.10) (Das *et al.*, 2022).

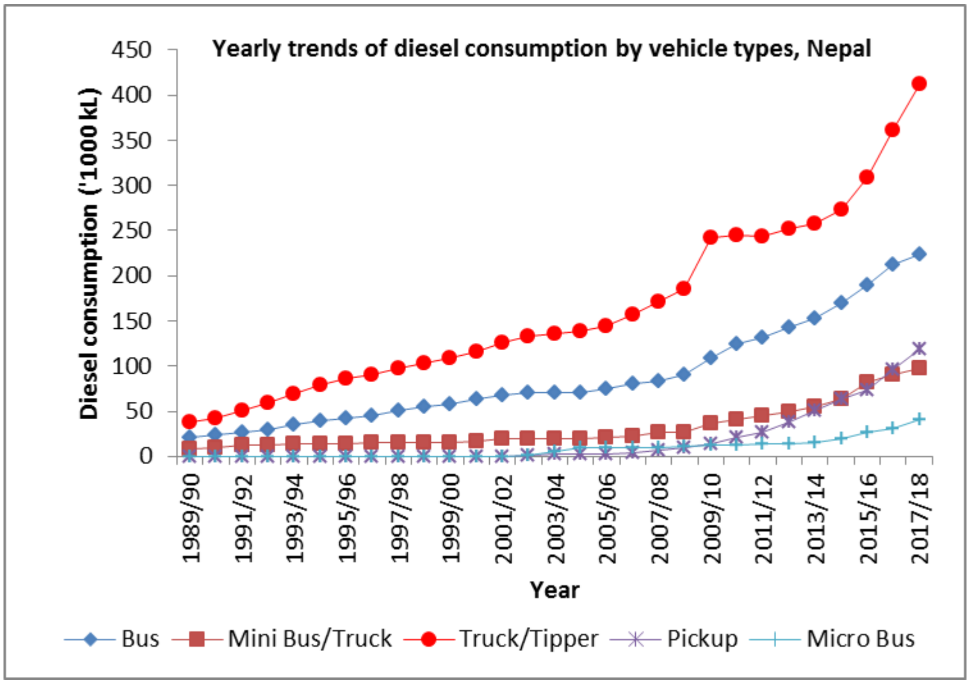

Figure 3.10: Status of diesel consumption in Nepal by vehicle category

3.4.4 Emission Factors

This study estimated the country-based EFs of pollutants from various diesel vehicles (Fig. 3.11; Appendix XIV-XV, XIX). The EFs were estimated as 2,559-2,605 for CO_2, 30.01-58.8 for CO, 1.03-1.59 for BC and 5.92-15.31 for $PM_{2.5}$ from buses, mini buses (2,271-2,574 for CO_2, 50-237 for CO, 0.6-2.2 for BC, and 1.6-5.4 for $PM_{2.5}$), mini truck/tipper (2,554-2,591 for CO_2, 39.6-62.5 for CO, 0.7-1.9 for BC, and 5.5-54.5 for $PM_{2.5}$), truck/tipper (2,564-2,602 for CO_2, 32.9-96.5 for CO, 0.4-2 for BC, and 3.5-16.3 for $PM_{2.5}$), pickup (2,596-2,610 for CO_2, 27.8- 35.8 for CO, 0.4-0.9 for BC, and 12.7-25.2 for $PM_{2.5}$), and micro bus (2,314-2,623 for CO_2, 19.7- 93.1 for CO, 0.4-1.3 for BC, and 2.1-7.3 for $PM_{2.5}$), in unit of g/L (Das *et al.*, 2022). Appendix XVII-XVIII, likewise, show EFs in Euro grade-wise based on emission measurement and calculation from collected studies.

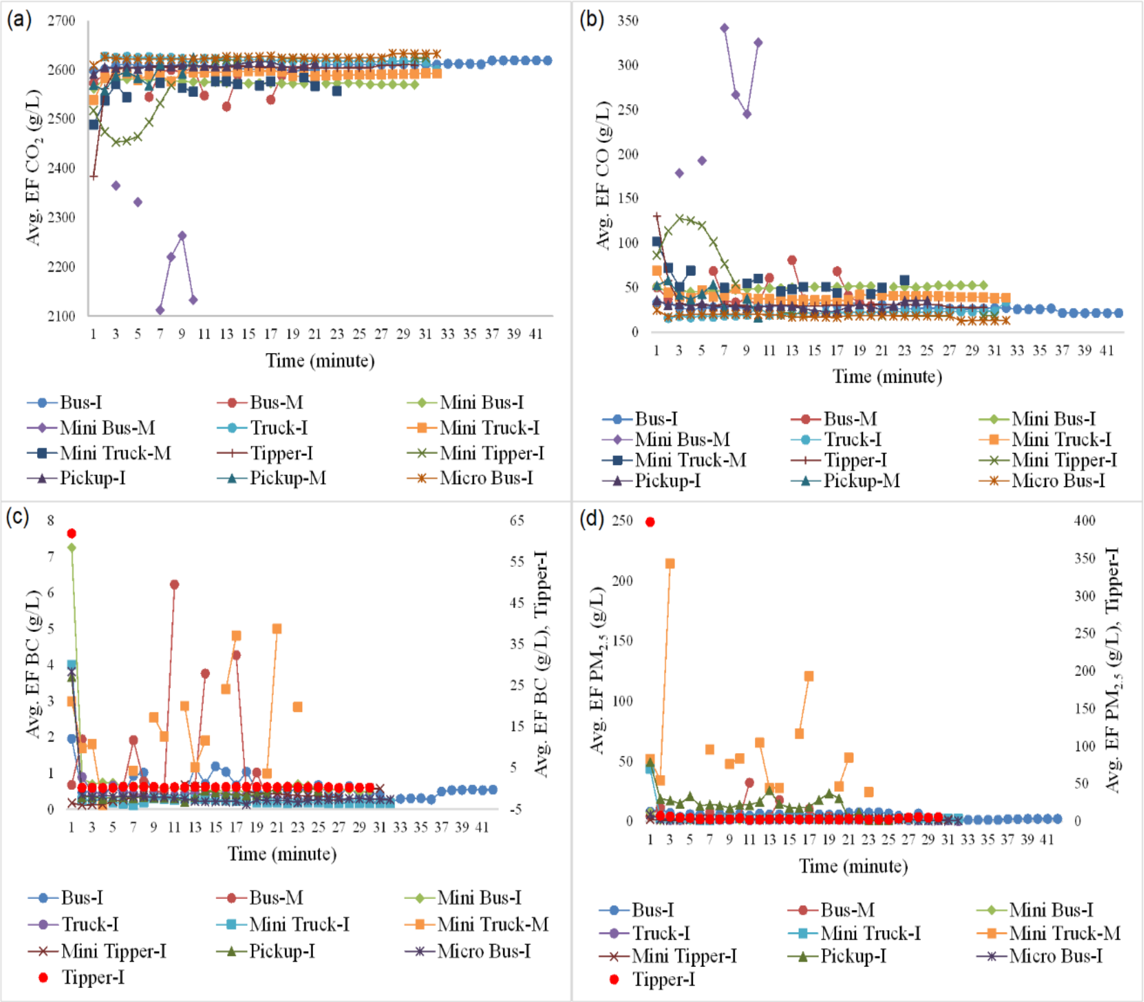

Figure 3.11: Emission Factors by vehicle category

The EFs were compared with other reported studies (Table 3.4). There are only limited studies that report fuel-based EFs for diesel vehicle categories. The EF of CO_2 of a diesel bus in idling condition was almost the same as Mool *et al.* (2020), a country-based reported value; however, CO and $PM_{2.5}$ were 1.5 and 1.7 folds lower, respectively, while BC was 1.5 folds higher than the reported literature. Likewise, the EF of CO_2, CO and $PM_{2.5}$ of diesel buses in moving condition was higher by 1.1, 3.8 and 13.1 folds, respectively, than the ARAI (2007), Indian-based value.

Although the EF of CO_2 was almost the same, the CO, BC, and $PM_{2.5}$ of trucks and tippers were higher by 1.2, 1.8, and 5.3 folds, respectively, than Mool *et al.* (2020). Compared with China, the EF of $PM_{2.5}$ was 8.3 folds higher than Cui *et al.* (2017) and BC was 2.9 folds higher than Wei *et al.* (2017). Likewise, for the USA, the EF of CO was 2 folds lower, while BC 2.1 and $PM_{2.5}$ were 11.2 folds higher (Park *et al.*, 2011). Compared with Park *et al.* (2011), the EFs of CO, BC and $PM_{2.5}$ of truck/tipper at moving conditions were higher by 3.9, 2.8 and 13.1 folds, respectively. However, CO_2 was 1.1 folds lower and CO 4.2 and $PM_{2.5}$ 3.5 folds higher than truck (HCV)-D, India (ARAI, 2007). CO, BC, and $PM_{2.5}$ levels were 2.7-3.9, 1.2, and 8.2-14 times lower in comparison to China (Wang *et al.*, 2011; Huo *et al.*, 2012). CO, BC, and $PM_{2.5}$ levels were also 3.5-3.6, 2.9, and 13.7 times lower than in the United States (Ban-weiss *et al.*, 2008; Burgard *et al.*, 2006; Bishop *et al.*, 2001). Germany reports an EF of CO_2 that is 1.1 folds higher than the estimates from this study, whereas BC for HDDT at moving conditions is 12.2 folds lower (Schneider *et al.*, 2008). Although the EF of CO_2 of the pickup at idling condition was almost the same as Mool *et al.* (2020), the EF of CO was higher by 1.3 folds, whereas BC and $PM_{2.5}$ were lower by 3.5 and 1.2 folds, respectively.

This study suggests variation in EFs as compared to other studies could be due to various vehicle categories and age-wise vehicle statistics (Park *et al.*, 2011), quality of fuel (Cui *et al.*, 2017; Wang *et al.*, 2011; Wei *et al.*, 2017), grade of road-way (Cui *et al.*, 2017), efficiency, and lowly driving situations (i.e., high and low speed) (Cui *et al.*, 2017; Wang *et al.*, 2011; Wang *et al.*, 2011; Wei *et al.*, 2017). Most of the diesel vehicles in the Kathmandu Valley are old and lack of proper maintenance (Ale & Nagarkoti, 2003; Mool *et al.*, 2020; Das *et al.*, 2018b). Low vehicle speeds inclusive of old and unmaintained vehicles with high mileage, coupled with narrow and hilly roads in the Kathmandu Valley. Pierson *et al.* (1996) carried out a prior tunnel investigation that examined the impact of roadway gradient and, subsequently, engine load on exhaust emissions. In comparison to downhill driving on a 0.6-3.8% grade, this study indicated that driving uphill on a 3.8 % grade nearly doubled the CO and NO_x EFs expressed per unit distance traveled and increased the VOC EF by 50% in the Fort McHenry Tunnel, USA. More recently, emissions in a Swedish tunnel with both uphill and downhill sections were demonstrated to depend on driving conditions, with EFs in grams per kilometer rising by a factor of up to 10 during congested when compared to smooth driving conditions (Sjodin, 1998).

Table 3.4: EFs of Nepal and the world (in g/kg).

References	Year	Vehicle category	CO_2	CO	BC	$PM_{2.5}$
This study	2019	Truck/Tipper-I, Nepal	3025	38.2	0.5	4.1
This study	2019	Truck/Tipper-M, Nepal	2982	113	2.7	19.2
Mool *et al.*, 2020	2017-2018	Tipper-I, Nepal	3137	33	0.4	6.9
ARAI, 2007	-	Truck (HCV)-D, India	3385	26.6	-	5.5
Cui *et al.*, 2017	-	HDDT-I, China	-	-	-	0.5
Wei *et al.*, 2017	2014	HDDT-I, China	-	-	0.2	-
Park *et al.*, 2011	2007	HDDT-I, USA	-	75	0.2	0.4
Mool *et al.*, 2020	2017-2018	Truck-I, Nepal	3101	56	1.3	37.1
Liu *et al.*, 2009	2007-2008	LDDT-M, China	-	-	-	0.6
Wang *et al.*, 2011	2009	HDDT-M, China	-	42	2.2	2.35
Huo *et al.*, 2012	2007 and 2011	HDDT-M, China	-	29	-	1.4
Subramanian *et al.*, 2009	2008	LDDT & HDDT-D, Thailand	-	-	-	8.4
Ban-weiss *et al.*, 2008	2008	HDDT-M, USA	-	-	0.9	1.4
Burgard *et al.*, 2006	2005	HDDT-M, USA	-	32	-	-
Bishop *et al.*, 2001	1997–1999	HDDT-M, USA	-	31	-	-
Schneider *et al.*, 2008	2005	HDDT-M, Germany	3190	-	0.2	-
This study	2019	Pick up-I, Nepal	3034	32.4	0.5	14.8
Mool *et al.*, 2020	2017-2018	Pick up-I, Nepal	3149	24	1.8	17.6
This study	2019	Pick up-M, Nepal	3019	41.6	1	29.2
ARAI, 2007	-	Bus (HCV)-D, India	2733	17.8		1.4
This study	2019	Bus-I, Nepal	3029	34.9	1.2	6.9
Mool *et al.*, 2020	2017-2018	Bus-I, Nepal	3107	52	0.8	11.9
This study	2019	Bus-M, Nepal	2976	68.4	1.8	17.8
This study	2019	Microbus-I, Nepal	3050	22.8	0.4	2.4
This study	2019	Microbus-M, Nepal	2690	108.3	0.4	8.5
Kirchstetter *et al.*, 1999	1997	HDDV-M , USA	-	-	1.3	2.5
Weingartner *et al.*, 1997	1993	HDDV-M, Switzerland	-	-	0.3	-
This study	2019	Mini Bus-I, Nepal	2994	58.2	0.7	1.8
This study	2019	Mini Bus-M, Nepal	2641	275.7	2.6	6.3
This study	2019	Mini Truck/Tipper-I, Nepal	3013	46	0.8	6.4
This study	2019	Mini Truck/Tipper-M, Nepal	2970	72.6	2.2	63.4
This study	2019	Mini Bus/Truck-I, Nepal	3050	22.8	0.4	2.4
This study	2019	Mini Bus/Truck-M Nepal	2690	108.3	1.5	8.5

Note: I=Idling condition, M=Moving condition

3.4.5 Emissions from diesel vehicles

On average, emission trends of CO_2, CO, BC, and $PM_{2.5}$ from diesel vehicles have marked an increase. In the year 2017/18, in the Bagmati zone/Kathmandu Valley, CO_2 emissions were estimated as 637-679 Gg, 9.7-26.2 Gg of CO, 0.2-0.8 Gg of BC, and 1.0-6.5 Gg of $PM_{2.5}$, higher by 16.8-17.3, 5.1-6.6, 4.5-4.9, and 6.6-9.6 folds, respectively, than the value for 1989/90 (Fig. 3.12–3.13; Appendix XX) (Das *et al.*, 2022). The emissions of diesel vehicles in the Bagmati zone/Kathmandu Valley for 2010/11 and 2014/15 were compared with total emissions (i.e., diesel and petrol) from the available literature (Table 3.5). The comparison showed the estimated value of CO_2 to be 1.4-1.5 and 2.2-folds lower than Shrestha *et al.* (2013b) and Ghimire & Shrestha (2014), respectively. Likewise, CO emissions were 0.6-2 and 0.4-1.2 fold lower than Shrestha *et al.* (2013b) and Ghimire & Shrestha (2014). Compared to Shrestha *et al.* (2013b), the values BC and $PM_{2.5}$ were 0.7-7 and 0.3-2.6 folds lower, respectively.

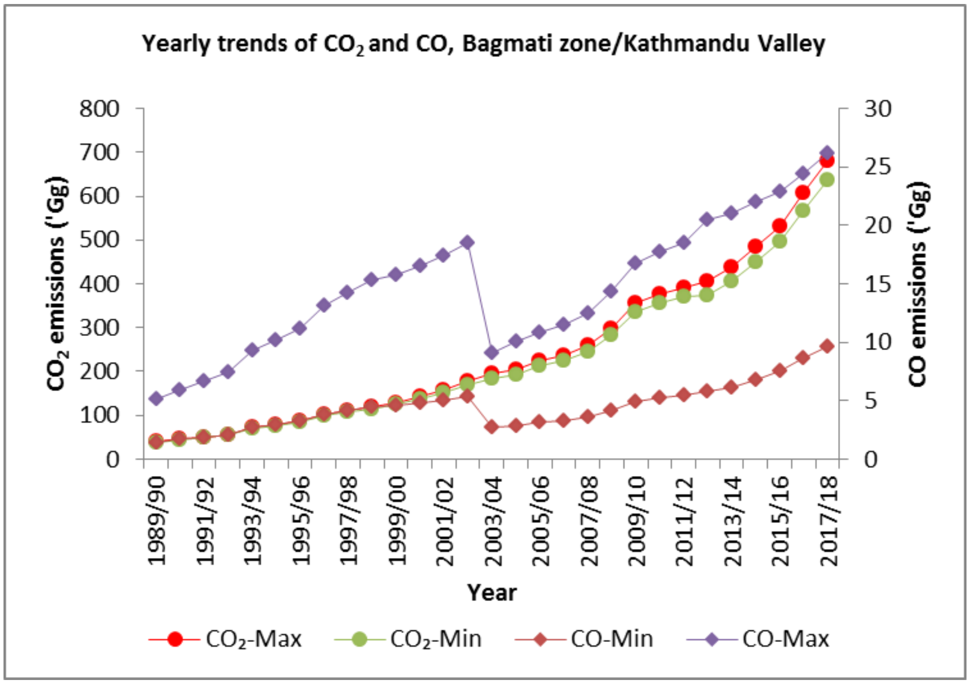

Figure 3.12: Status of CO_2 and CO emission in Bagmati zone/Kathmandu Valley

The CO_2 emissions in the year 2017/18 for Nepal were estimated as 2214-2781 Gg, 27.7-88.8 Gg of CO, 0.5-3.6 Gg of BC, and 3.4-23.5 Gg of $PM_{2.5}$, higher by 13.1-16, 5.1-5.3, 3-4.9, and 5.9-7.5 times, respectively, than the value for 1989/90 (Fig. 3.14-3.15; Appendix XXI) (Das *et al.*, 2022). The diesel vehicle emissions from 2008/09 were compared with total emissions (i.e., diesel and petrol) from the transport sector from Sadavarte *et al.* (2019). The results suggest CO_2 emission was very close; however, CO and BC were 1.7-6.4 and 0.4-4.5 folds respectively lower, while $PM_{2.5}$ emission ranged in between 1.2-10.9 folds with that of the literature (Table 3.5).

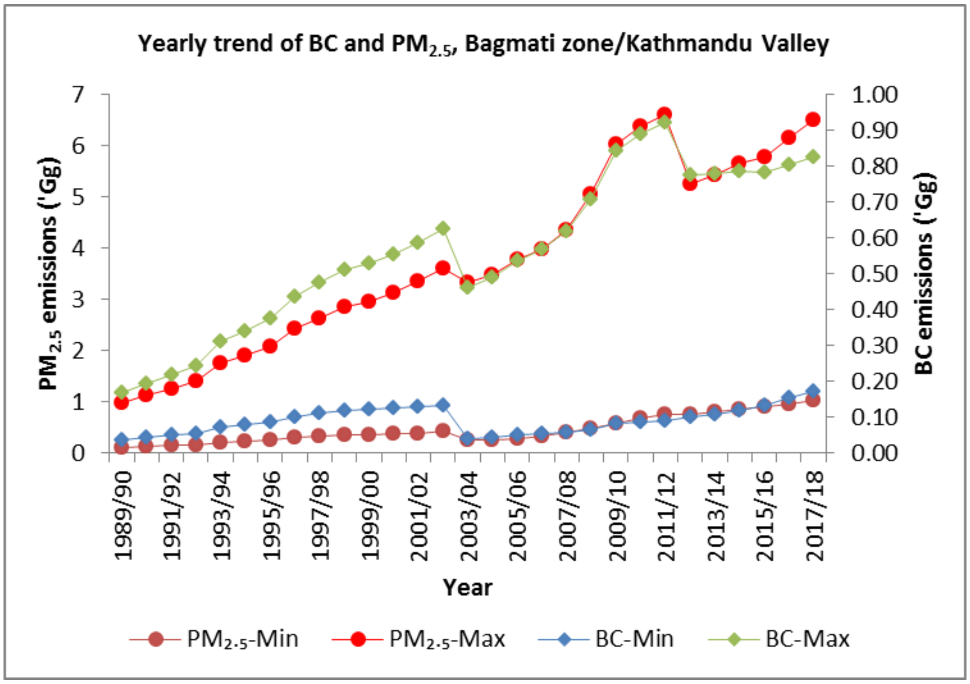

Figure 3.13: Status of BC and $PM_{2.5}$ emission in Bagmati zone/Kathmandu Valley

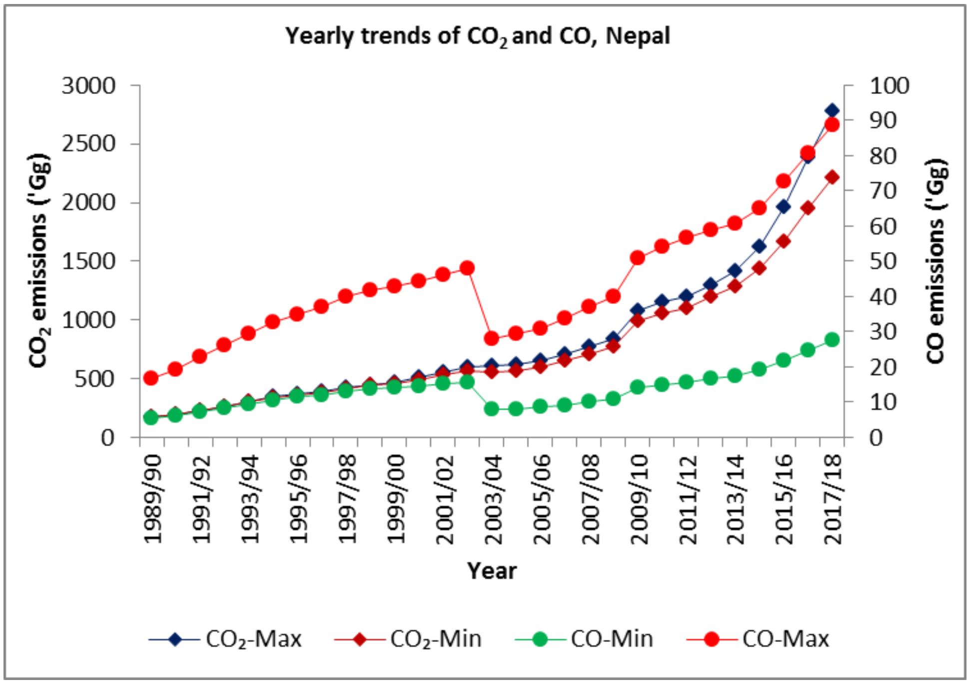

Figure 3.14: Status of CO_2 and CO emission in Nepal

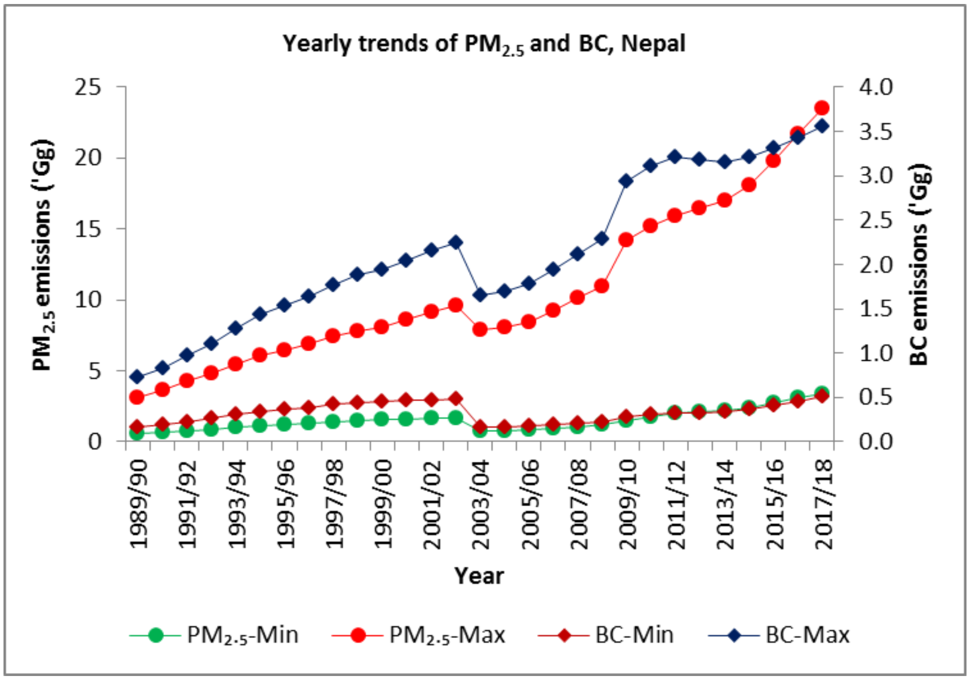

Figure 3.15: Status of BC and $PM_{2.5}$ emission in Nepal

The differences in the findings were due to dissimilarities in choosing parameters such as vehicle types, age-wise vehicle statistics, vehicles' survival fraction, VKT, fuel utilization, and efficiency. Moreover, EFs that were considered by the above-reported literature (e.g., ARAI, 2007; Jaiprakash, *et al.*, 2016; Jayarathane *et al.*, 2018; Kim-Oanh *et al.*, 2010; Sadavarte & Venkataraman, 2014; Shrestha *et al.*, 2013; Stockwell *et al.*, 2016; Wu *et al.*, 2015; Yang *et al.*, 2019) were entirely secondary-based values and not country-specific, which could have affected the results.

Table 3.5: Comparative estimates of emissions (in Gg).

Nepal	Year	CO_2	CO	BC	$PM_{2.5}$
This study	2008/09	770-840	10.9-39.9	0.2-2.3	1.2-10.9
Sadavarte et al., 2019	2008	782	69.37	0.89	1.8
Kathmandu Valley					
This study	2010/11	1,057-1,150	15.0-54.2	0.3-3.1	1.8-15.2
Shrestha et al., 2013	2010	1,554	31	2.117	4.685
This study	2014/15	1,441-1,628	19.1-65.1	0.4-3.2	2.4-18.1
Ghimire & Shrestha, 2014	2014	3,196	23.81	-	-

3.4.5.1 CO_2 emissions

In the year 1989/90 in the Bagmati zone/Kathmandu Valley, the highest contribution of CO_2 emissions was from trucks/tippers (32.1-32.7%), followed by buses (26.2-26.5%), mini buses (24.3-24.9%), and mini trucks/tippers (16.5-16.9%). In the year 2017/18, the CO_2 emissions share of buses, mini buses, trucks/tippers, and mini trucks/tippers declined with respect to 1989/90; however, the share of pickups and micro buses increased significantly. The highest contribution of CO_2 was from the bus (21.7-22.2%), followed by truck/tipper (21.3-21.5%), mini bus (17.4-19.6%), pickup (18.2-18%), mini truck/tipper (11.7-12.3%) and micro bus (7.2-7.8%) (Fig. 3.16) (Das *et al.*, 2022).

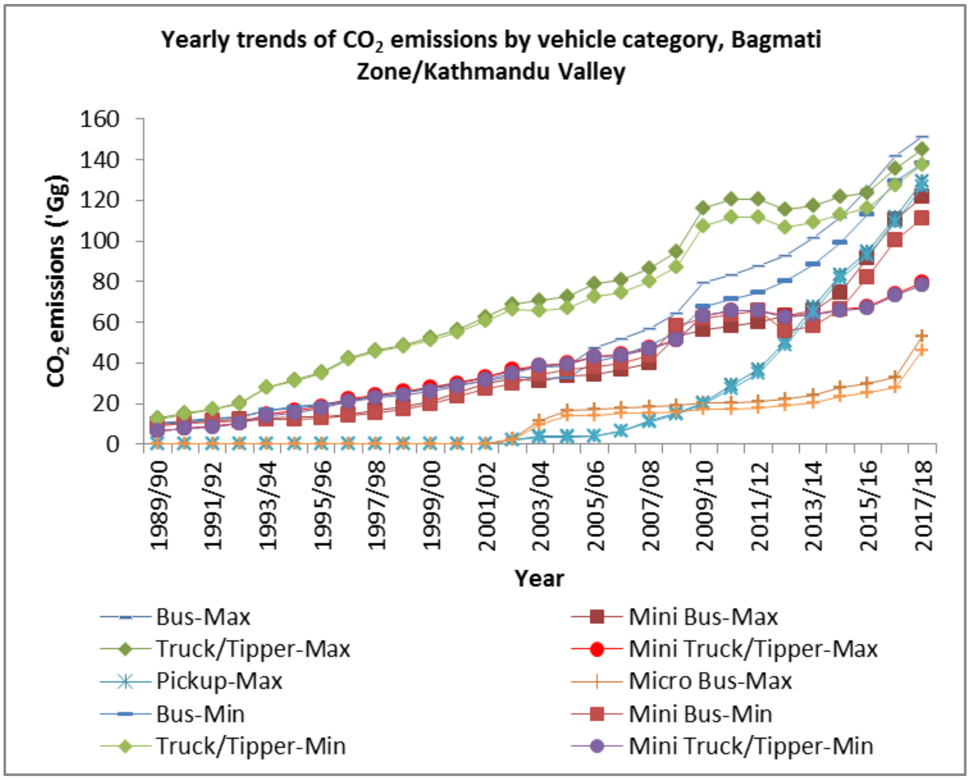

Figure 3.16: Status of CO_2 emission in Bagmati zone/Kathmandu Valley by vehicle category

Likewise, in the year 1989/90 in Nepal, the highest contribution of CO_2 emissions was from trucks/tippers (56.4-56.8%), followed by buses (31.4-31.5%), and mini bus/trucks (11.7-12.2%). In the year 2017/18, the CO_2 emissions share of trucks and tippers slightly declined with respect to 1989/90. However, it still remained the highest contribution (46.3-54.9%) followed by buses (21-24.2%), pickups (11.1-13.7%), mini bus/trucks (9.2-11.6%), and micro buses (3.8-4.2%).

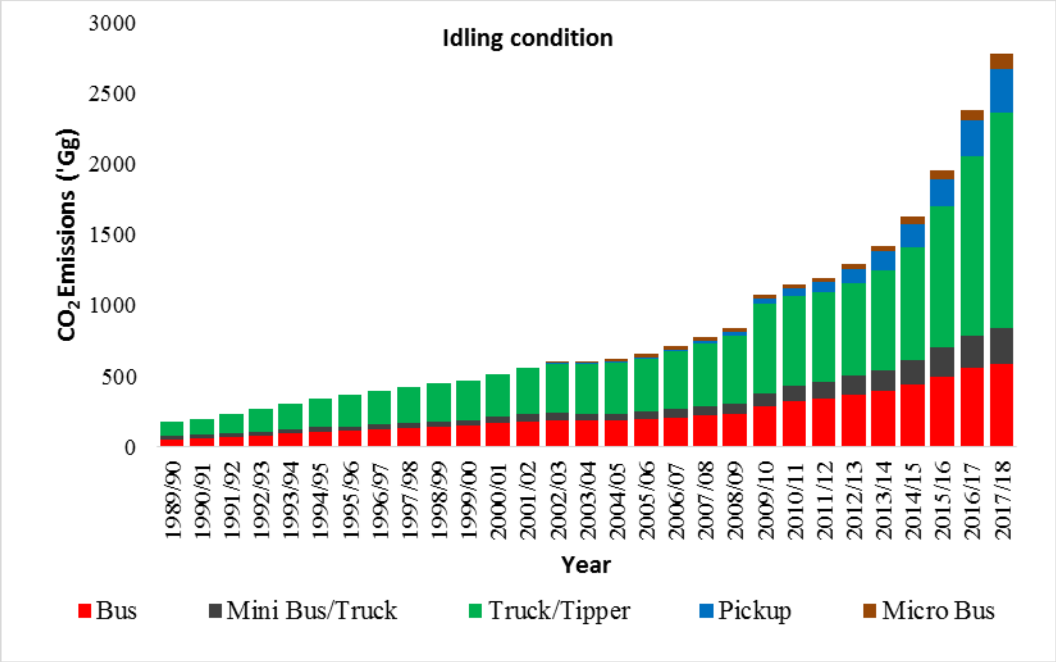

Figure 3.17: Status of CO_2 emission in Nepal by vehicle category (Idling condition)

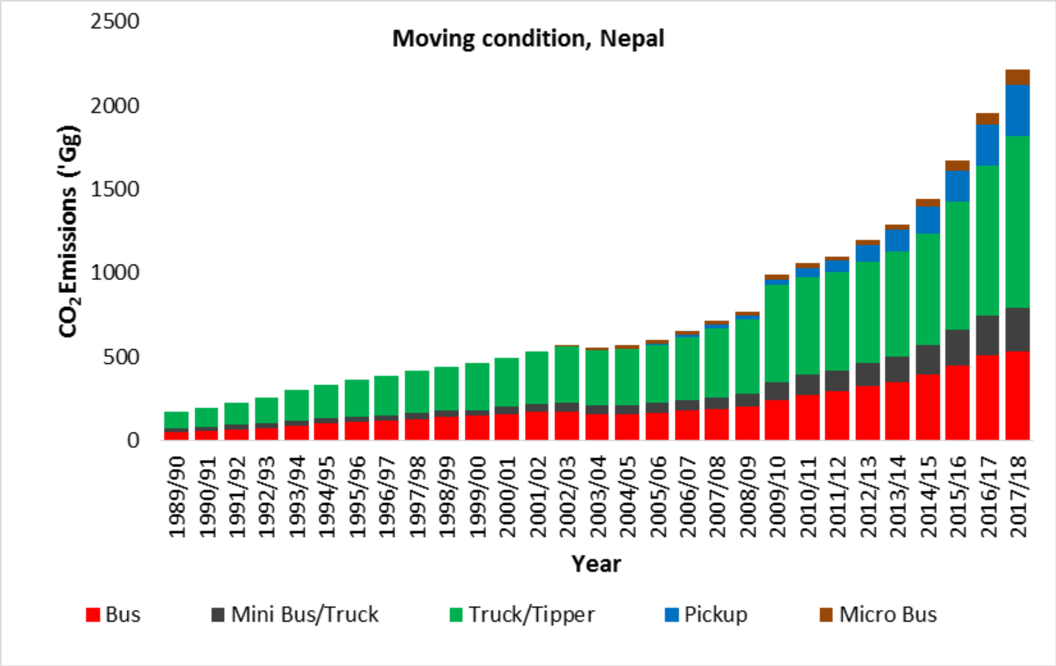

Figure 3.18: Status of CO_2 emission in Nepal by vehicle category (Moving condition)

Although the pickup and microbus were first introduced in Nepal in 2002/03, the vehicle numbers had increased to 96.3 times and 33 times, respectively, by 2017/18. Whereas, minibus/trucks in 2017/18 rose 12.4 times more than in 1989/90. Due to higher registration growth, pickups' CO_2 emissions topped those of mini buses/trucks and microbuses (Fig. 3.17-3.18) (Das *et al.*, 2022).

3.4.5.2 CO emissions

In the year 1989/90 in the Bagmati zone/Kathmandu Valley, the highest contribution of CO emissions was from mini buses (35.2-39.8%), followed by trucks/tippers (27.2-31.5%), mini trucks/tippers (23.9-27%), and buses (6-9.4%). In the year 2017/18, the CO emissions share of buses was almost the same; however, mini buses, truck/tippers, and mini trucks/tippers declined with respect to 1989/90. The share of pickups and micro buses increased significantly. The mini bus contributed the most CO (24.4-31.7%), followed by the truck/tipper (19.2-24.3%), the bus (14.9-18%), the micro bus (4.8-16.5%), the pickup (13.1-13.6%), and the mini truck/tipper (6.9-12.9%) (Fig. 3.19) (Das *et al.*, 2022).

Nationally, in the years 1989/90, the highest contribution of CO emissions was from trucks and tippers (64.2-66.1%), followed by mini buses and trucks (20.4-26.2%) and buses (9.6-13.4%). In the year 2017/18, the CO emissions share of buses, mini buses, mini trucks, and trucks/tippers declined with respect to 1989/90; however, the share of pickups and micro buses increased significantly. Trucks/tippers contributed the most CO (48.9-51%), followed by buses (16.8-24%), mini buses/trucks (12.5-15%), pickups (8.3-11.1%), and micro buses (3.4-8.8%) (Fig. 3.20-3.21) (Das *et al.*, 2022).

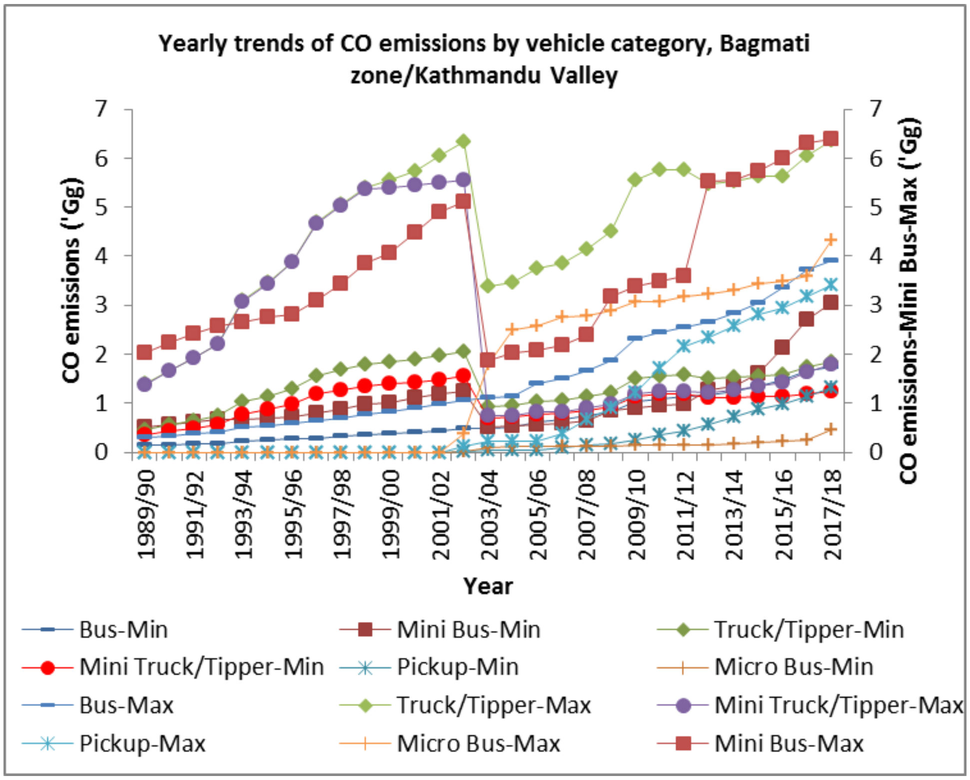

Figure 3.19: Status of CO emission in Bagmati zone/Kathmandu Valley by vehicle category

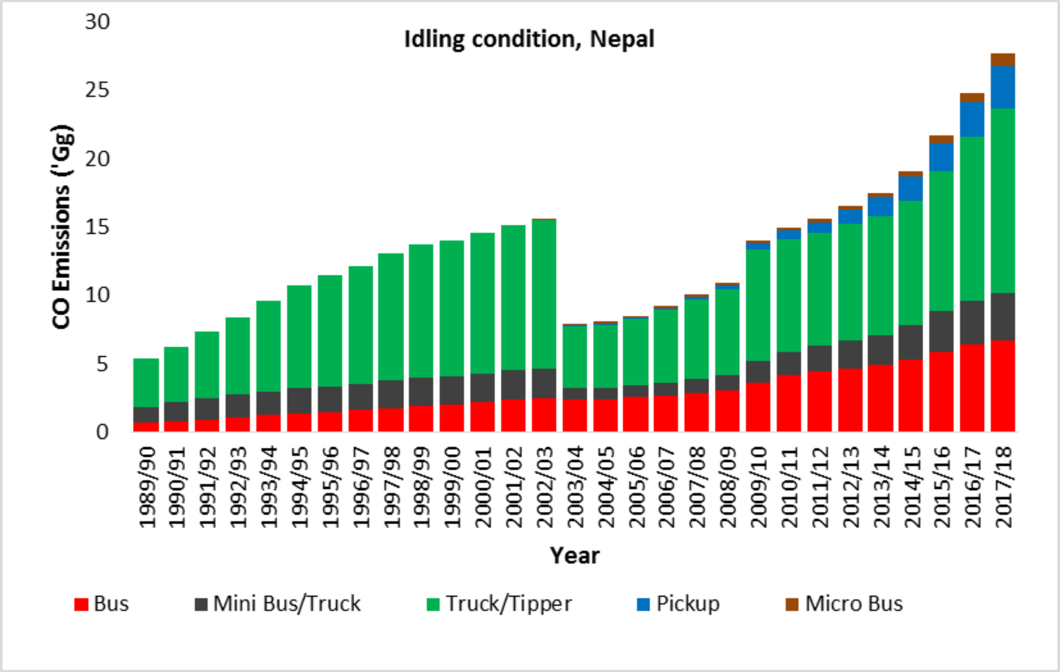

Figure 3.20: Status of CO emission in Nepal by vehicle category (Idling condition)

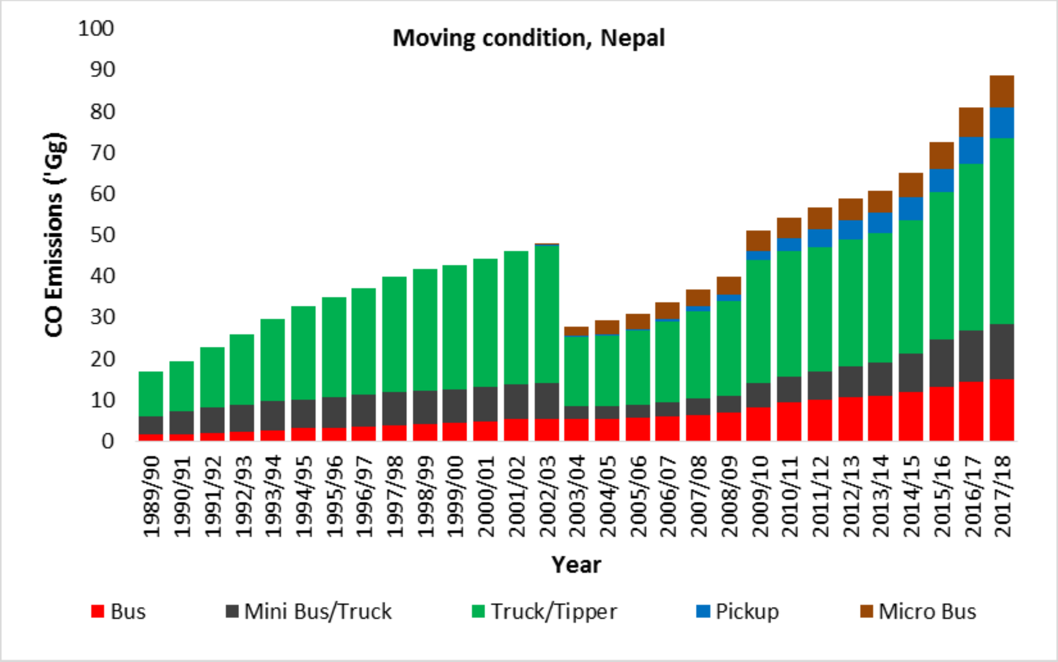

Figure 3.21: Status of CO emission in Nepal by vehicle category (Moving condition)

3.4.5.3 BC emissions

In the year 1989/90 in the Bagmati zone/Kathmandu Valley, the highest contribution of BC emissions was from truck/tipper (33.7-36.5%), followed by mini bus (26.7-26.8%), bus (18.6-21.3%), and mini truck/tipper (18.2%). Bus, minibus, truck/tipper, and mini truck/mini tipper traffic decreased in 2017/18 compared to 1989/90. The share of pickups and micro buses increased significantly. Trucks/tippers (12.6-37.6%) contributed the most, followed by buses (26.6-32.3%), mini buses (6.8-29.3%), pickups (12.2-14.2%), micro buses (5.9-9.3%) and mini truck/tippers (5.7-7.6%) (Fig. 3.22) (Das *et al.*, 2022).

Likewise, in the year 1989/90 in Nepal, the highest contribution of BC emissions was from trucks/tippers (60.5-64.5%), followed by buses (22.4-26.1%), and mini bus/trucks (13.1-13.4%). The share of BC emissions from buses, mini buses, mini trucks, and trucks/tippers decreased in 2017/18 compared to 1989/90; however, the share of pickups and micro buses increased significantly. Truck/tipper contributed the most BC (34.3-55%), followed by bus (23.1-41.3%), mini bus/truck (8.1-12.1%), pickup (6-12%), and micro bus (3.8-4.3%) (Fig. 3.23-3.24) (Das *et al.*, 2022).

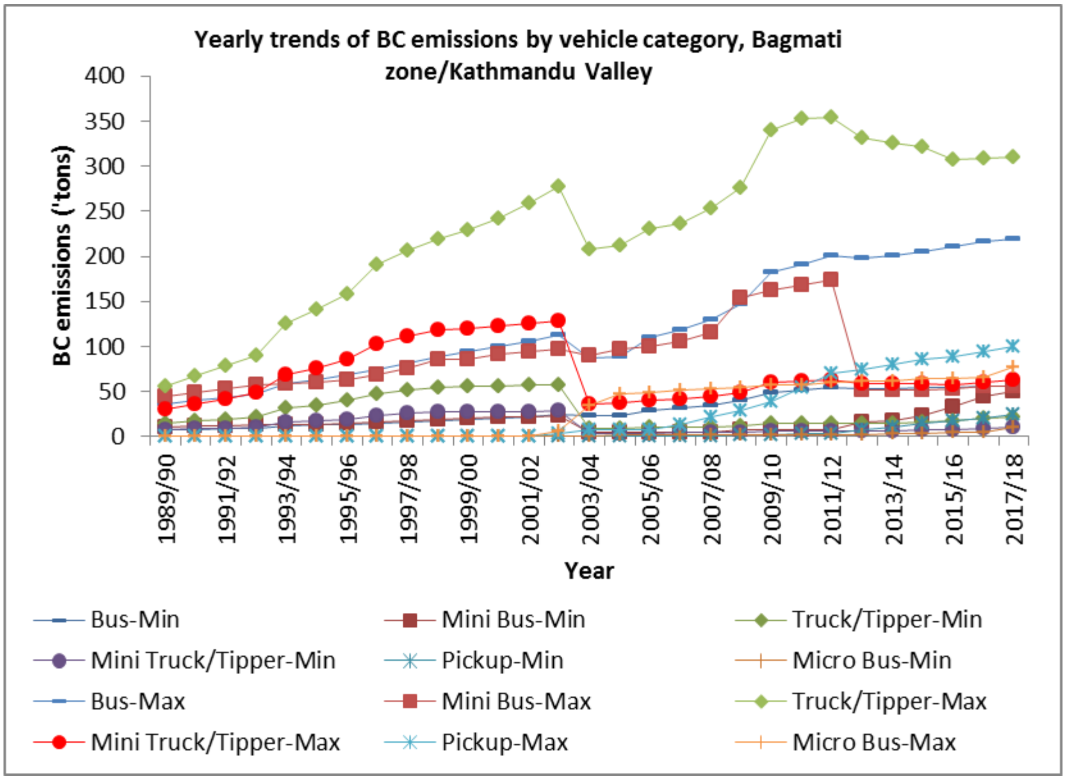

Figure 3.22: Status of BC emission in Bagmati zone/Kathmandu Valley by vehicle category

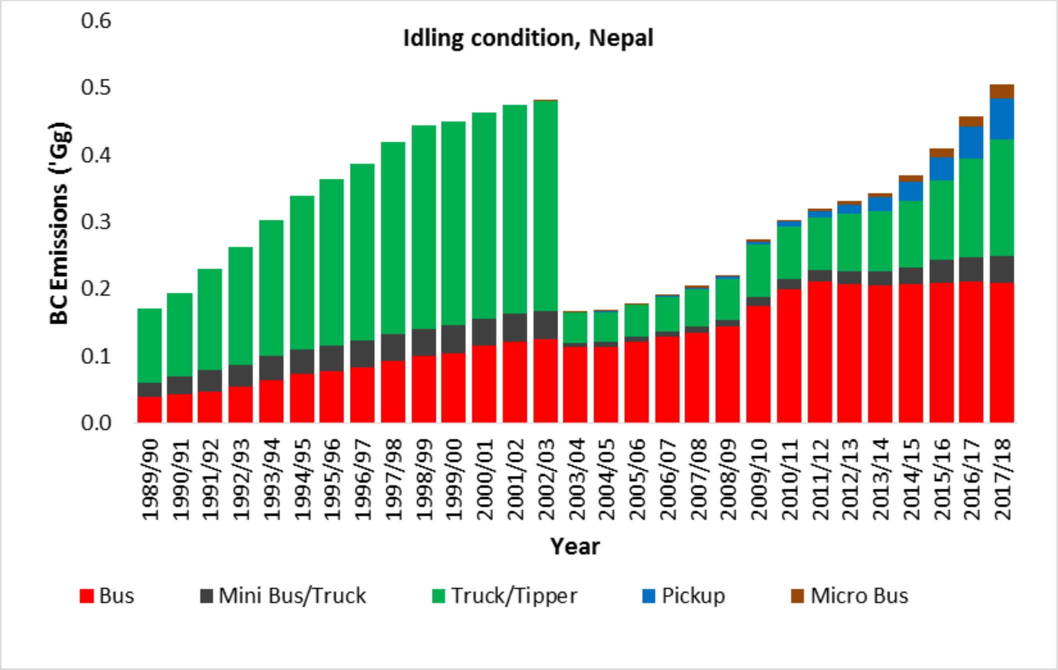

Figure 3.23: Status of BC emission in Nepal by vehicle category (Idling condition)

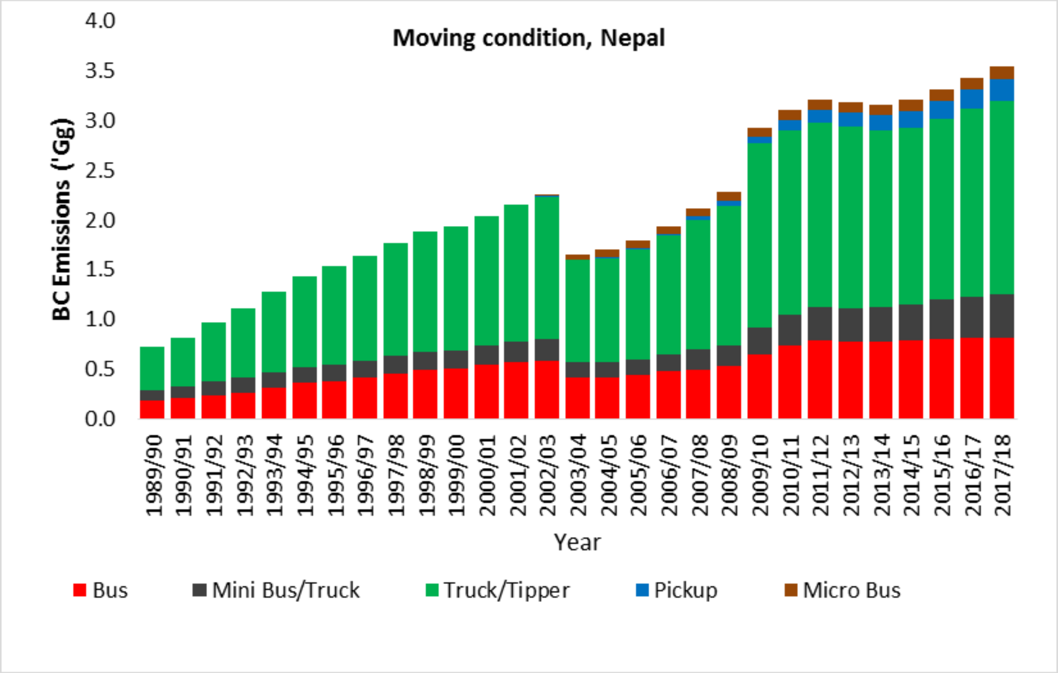

Figure 3.24: Status of BC emission in Nepal by vehicle category (Moving condition)

3.4.5.4 $PM_{2.5}$ emissions

In the year 1989/90 in the Bagmati zone/Kathmandu Valley, the highest contribution of $PM_{2.5}$ emissions was from truck/tippers (20.2-37.8%), followed by mini buses (18.7-38.4%), buses (15.4-30.8%), and mini trucks/tippers (12.7-26%). Bus, mini bus, truck/tipper, and mini truck/tipper decreased in 2017/18 compared to 1989/90. The share of pickups and micro buses increased significantly. Pickups contributed the most $PM_{2.5}$ (22.6-41.5%), followed by buses (20.4-31.4%), mini trucks/tippers (6.3-28.5%), truck/tippers (10.8-22.6%), mini buses (3.1-6.2%), and micro buses (2.8-3.7%) (Fig. 3.25) (Das *et al.*, 2022).

Nationally, in the years 1989/90, the highest contribution of $PM_{2.5}$ emissions was from mini buses/trucks (8.1-25.8%), followed by trucks/tippers (48.9-59.9%), and buses (25.4-32.9%). In the year 2017/18, the $PM_{2.5}$ emissions share of buses, minibus/trucks, and truck/tippers declined with respect to 1989/90; however, the share of pickups and micro buses increased significantly. Trucks/tippers contributed the most $PM_{2.5}$ (28.8-42.2%), followed by pickups (14.5-26.7%), buses (21.4-36.2%), mini buses/trucks (6.1-20.3%), and micro buses (1.5-2.2%) (Fig. 3.26-3.27).

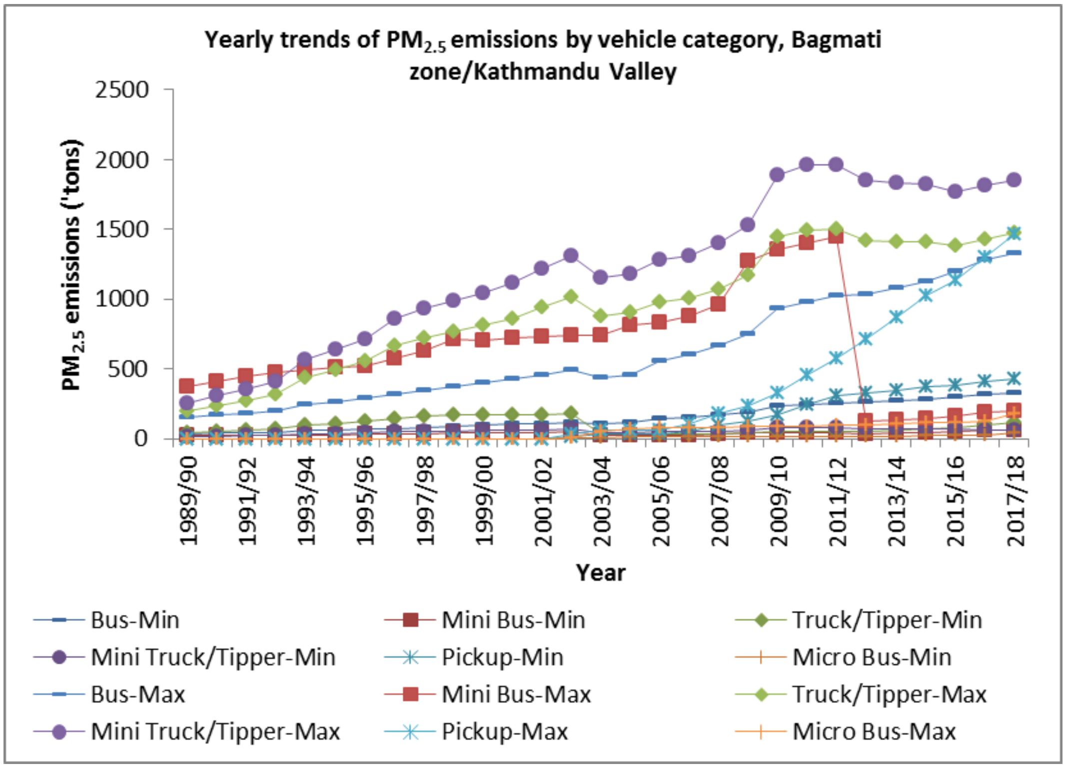

Figure 3.25: Status of $PM_{2.5}$ emissions in Bagmati zone/Kathmandu Valley by vehicle category

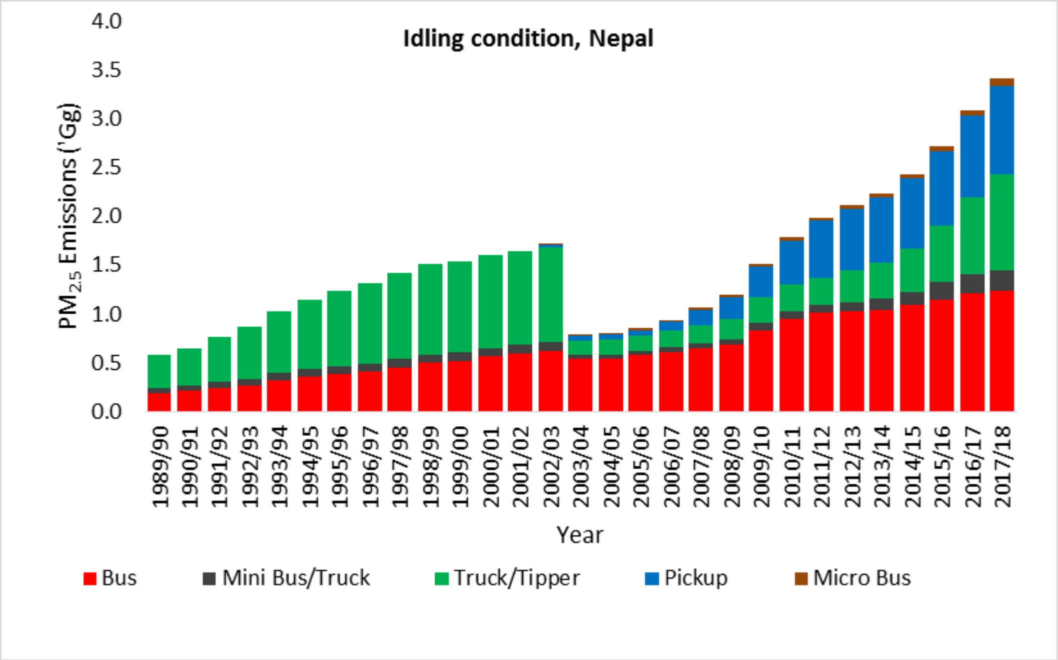

Figure 3.26: Status of $PM_{2.5}$ emissions in Nepal by vehicle category (Idling condition)

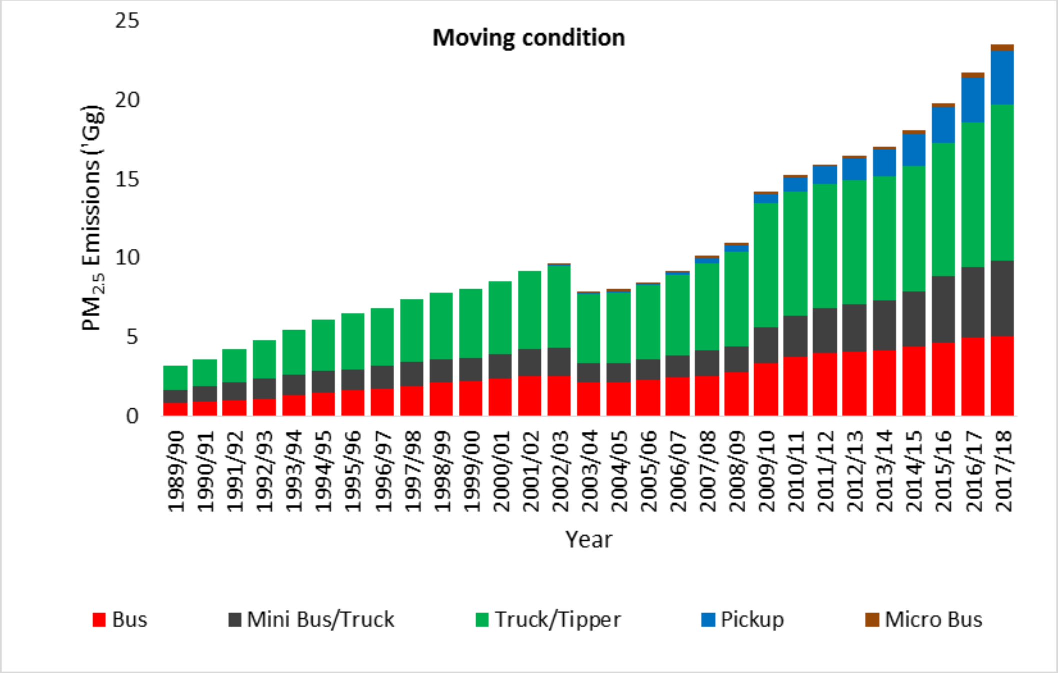

Figure 3.27: Status of $PM_{2.5}$ emissions in Nepal by vehicle category (Moving condition)

As there was no registration of pickups and micro buses in 1989/90 in the Kathmandu Valley and Nepal, CO_2, CO, BC, and $PM_{2.5}$ emissions from both vehicle categories were estimated at zero.

3.4.6 Daily emissions by road types in Bagmati zone/Kathmandu Valley

The total daily diesel vehicle emissions from outside ring road, ring road and inside ring road in the year 2017/18 in the Bagmati zone/Kathmandu Valley were estimated as CO_2 (1746-1861 tons), CO (26.4-71.8 tons), BC (0.5-2.3 tons), and $PM_{2.5}$ (2.8-17.8 tons). The emission of CO_2 in outside ring road in the year 2017/18 was estimated to be 958-1016 tons, higher by 1.7 folds (ring road) and 3.9-4 folds (inside ring road). The CO emission (14.78-39.30 tons) was higher by 1.8-1.9 folds (ring road) and 3.5-4.3 folds (inside ring road). Likewise, BC (0.24-1.26 tons) was higher by 1.5-1.7 folds (ring road) and 3.6-4.6 folds (inside ring road). Finally, $PM_{2.5}$ (1.45-10.23 tons) was higher by 1.5-1.8 folds (ring road) and 3.2-5.1 folds (inside ring road) (Fig. 3.28-3.29).

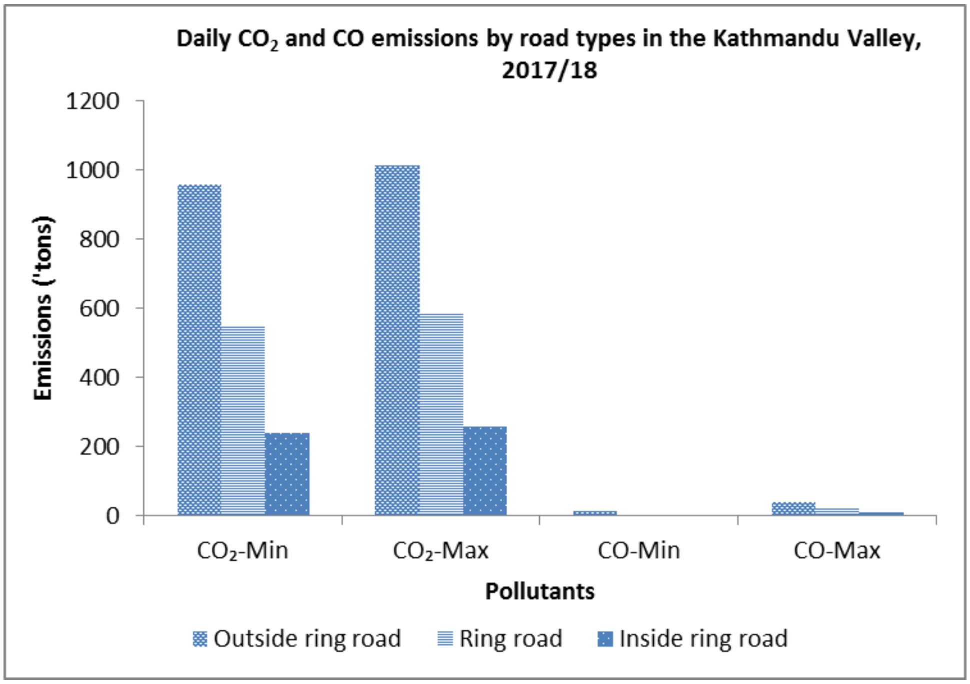

Figure 3.28: CO and CO_2 emission by road types in Bagmati zone/Kathmandu Valley

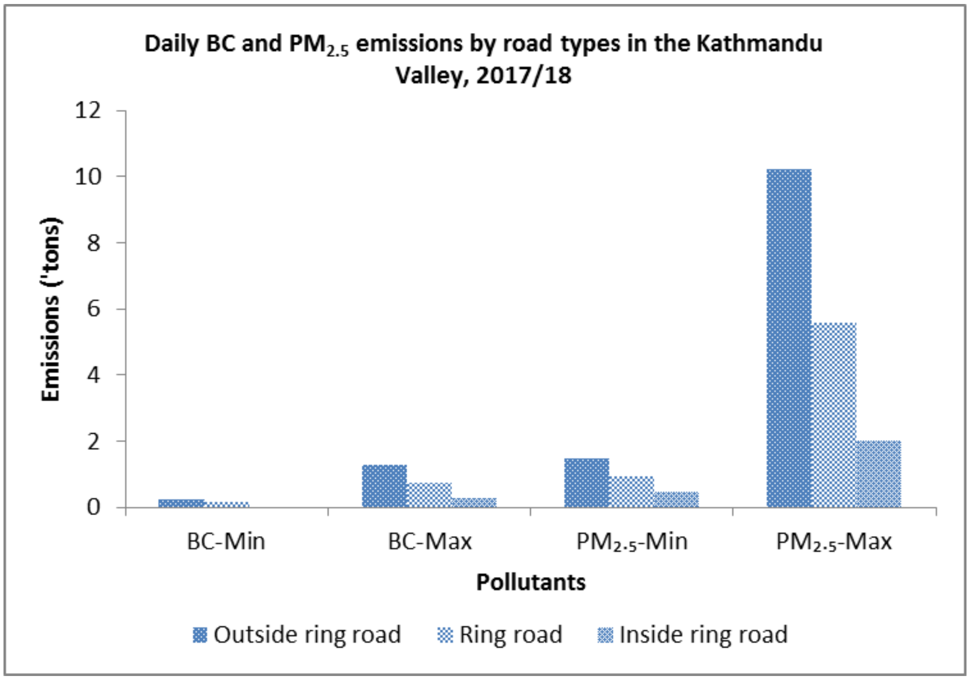

Figure 3.29: BC and $PM_{2.5}$ emission by road types in Bagmati zone/Kathmandu Valley

3.4.6.1 Outside ring road

Total contribution of CO_2 in the outside ring road by truck/tipper was 28.1-28.2%, mini bus (18.6-20.3%), pickup (18-18.7%), bus (14.5-14.9%), mini truck/tipper (13.9-14.6%), and micro bus (5.1-5.6%). Likewise, CO contribution by mini bus was 25.3-32.1%, truck/tipper (24.6-31.9%), mini truck/tipper (8.2-15%), pick up (12.9-13.1%), bus (10-11.8%), and micro bus (3.3-11.8%) (Fig. 3.30-3.31). BC contribution by truck/tipper was 17.7-48.5%, mini bus (6.9-32.4%), bus (17.5-23.1%), pickup (11.8-15%), mini truck/tipper (7.3-8.9%), and micro bus (4.5-6.5%). Lastly, $PM_{2.5}$ contribution from mini truck/tipper was (8.1-32.3%), truck/tipper (15.3-28.4%), pickup (13-22.7%), bus (13-22.7%), mini bus (3.1-6.9%), and micro bus (1.9-2.9%) (Fig. 3.32-3.33).

3.4.6.2 Ring road

Total contribution of CO_2 in the ring road by bus was 35.5-36.2%, truck/tipper (16.7-17%), mini bus (15.8-16.2%), pickup (14.2-14.9%), mini truck/tipper (11-11.6%), and micro bus (5.3-5.7%). Likewise, CO contribution by bus was 25.9-29.7%, mini bus (25.9-29%), truck/tipper (15.2-20.4%), mini truck/tipper (6.9-12.3%), micro bus (3.5-12.8%), and pickup (10.3-10.4%) (Fig. 3.30-3.31). BC contribution by bus was 42.5-48.1%, truck/tipper (9.1-29%), mini bus (6-24.2%), pickup (9.7-8.9%), mini truck/tipper (4.9-7%), and micro bus (3.9-6.6%). Lastly, $PM_{2.5}$ contribution by bus was 33.4-48.8%, pickup (17-29.6%), mini truck/tipper (5.6-26.9%), truck/tipper (8.1-17.9%), mini bus (2.8-5.4%), and micro bus (2.1-2.6%) (Fig. 3.32-3.33).

3.4.6.3 Inside ring road

Total contribution of CO_2 in the inside ring road by pickup was 30.6-32.3%, micro bus (19.9-21.3%), bus (19.1-19.3%), mini bus (18.7-19.2%), truck/tipper (5.2-5.3%), and mini truck/tipper (4.4-4.7%). Likewise, CO contribution by micro bus was 13.9-39.7%), mini bus (22.9-36%), pickup (18.5-23.3%), bus (11.4-16.7%), mini truck/tipper (2.3-5.2%), and truck/tipper (5-5.2%) (Fig. 3.30-3.31). BC contribution by mini bus was 8.3-30.2%, micro bus (15.6-29.1%), bus (26.4-27.1%), pickup (21.1-22.5%), truck/tipper (3-10.5%) and mini truck/tipper (2.1-3.3%). Lastly, $PM_{2.5}$ contribution from pickup was 44.6-57.6%, bus (21.8-23.6%), micro bus (8.8-9.5%), truck/tipper (2.3-6.8%), and mini bus (4.1-5.7%) (Fig. 3.32-3.33).

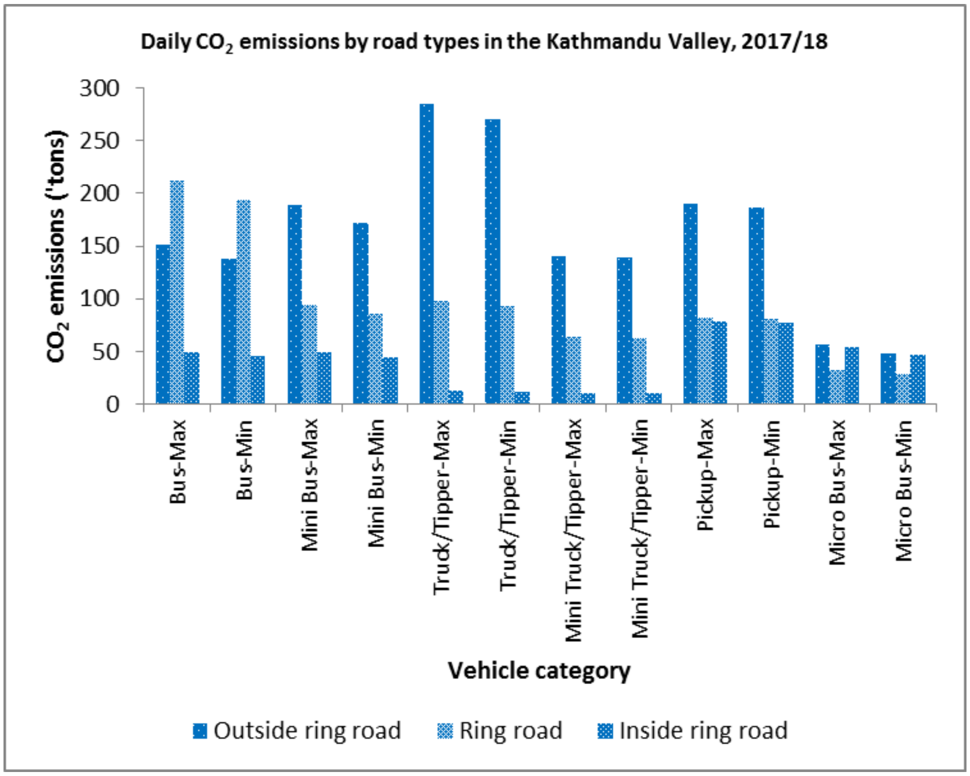

Figure 3.30: CO_2 emission by road types in the Bagmati zone/Kathmandu Valley

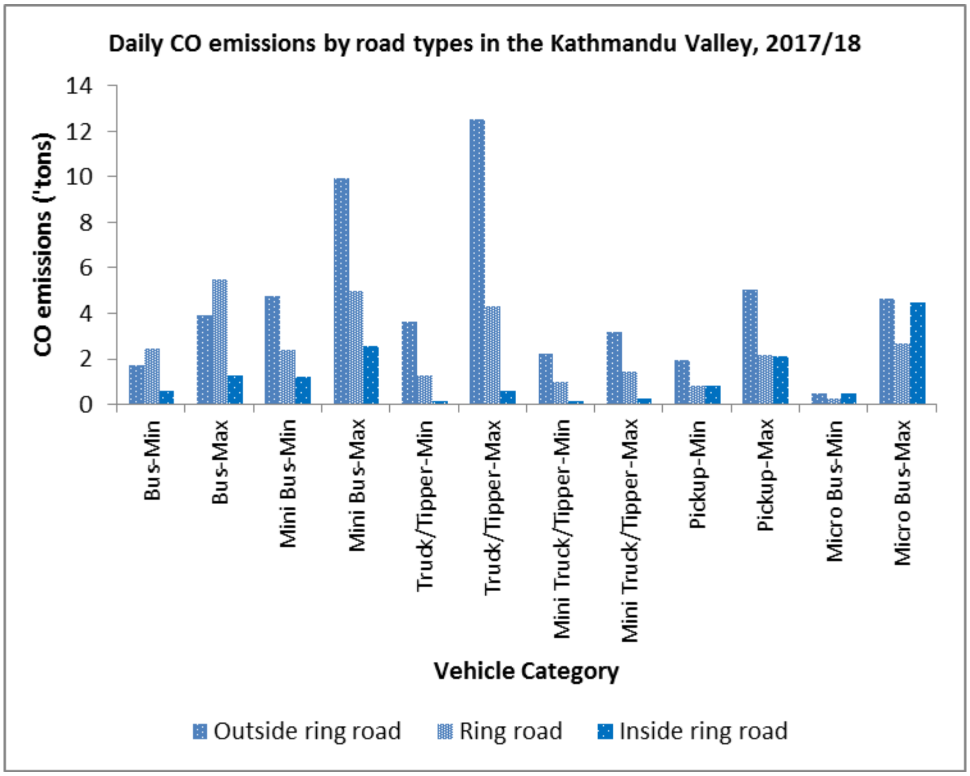

Figure 3.31: CO emission by road types in the Bagmati zone/Kathmandu Valley

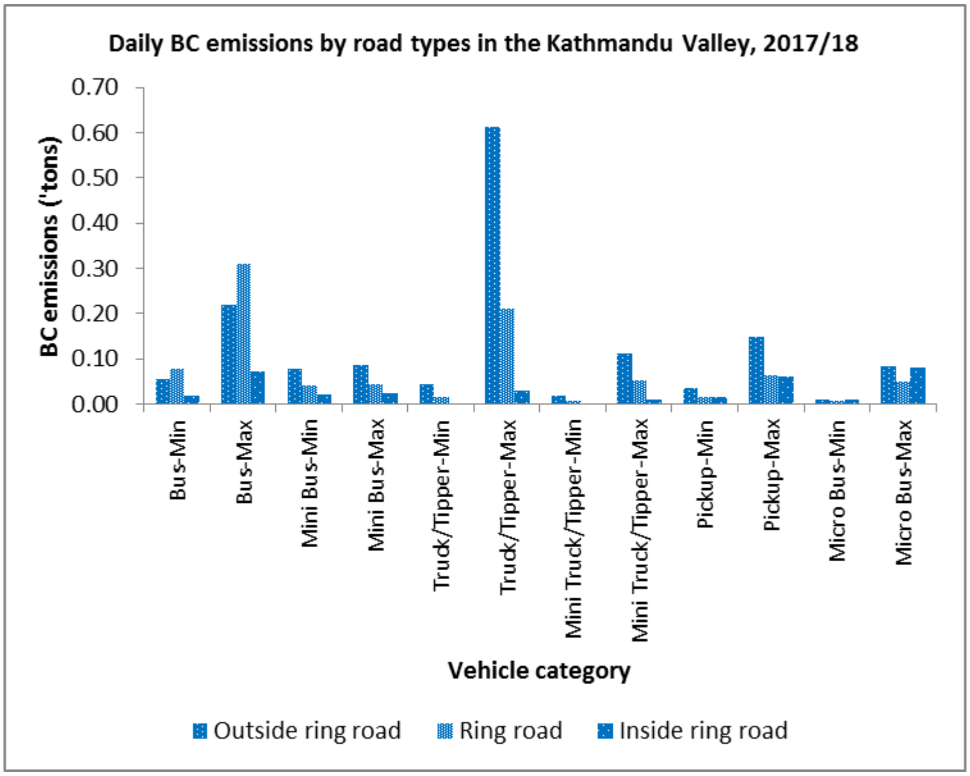

Figure 3.32: BC emission by road types in the Bagmati zone/Kathmandu Valley

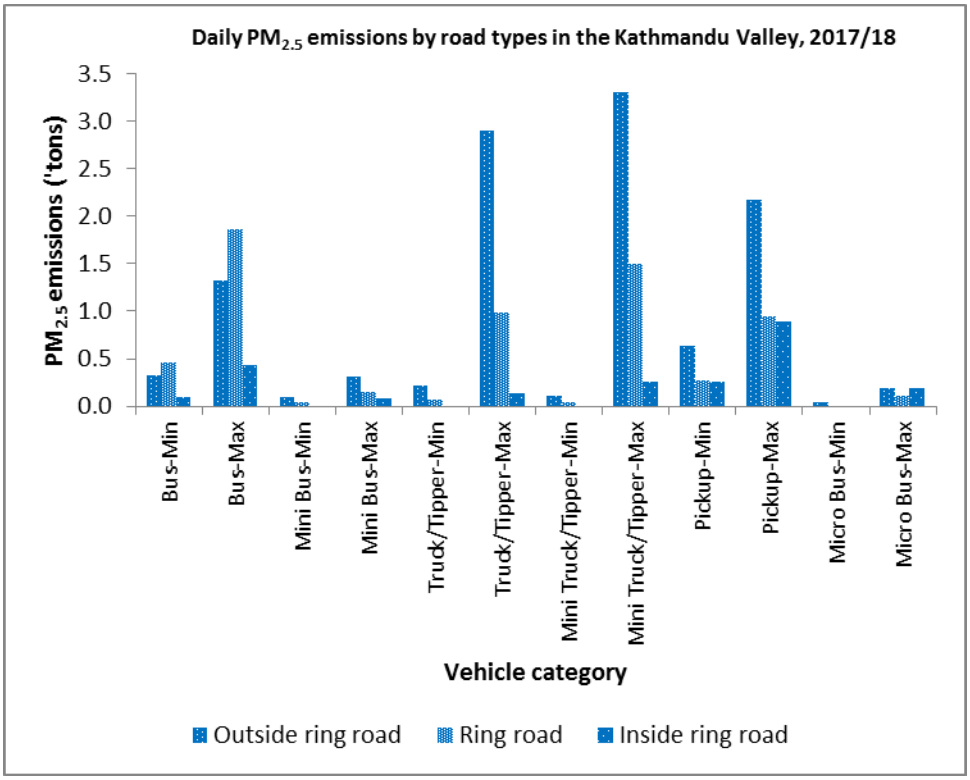

Figure 3.33: $PM_{2.5}$ emission by road types in the Bagmati zone/Kathmandu Valley

3.4.7 Daily temporal emissions

The study suggests each pollutant varies differently in a timely manner. The daily highest CO_2 emissions (138.3-147.2 tons) occurred at 2-3 pm in the Bagmati zone/Kathmandu Valley for the year 2017/18. The share of CO_2 emissions from diesel vehicles at this time was by pickup (23.5-24.5%), bus (20.6-21.2%), truck/tipper (17.6-17.7%), mini bus (13.5-13.9%), mini truck/tipper (12.4-13%), and micro bus (10.6-11.5%). In contrast, the highest CO (2-5.8 tons) was at 11 am - 12 pm (Das *et al.*, 2022). The total contribution of CO emissions at this time was by mini bus (20-27.9%), micro bus (7.8-24.9%), truck/tipper (18.9-22.3%), bus (13.5-17.6%), pickup (12.5-13.9%), and mini truck/tipper (6.9-13.9%) (Fig. 3.34-3.35).

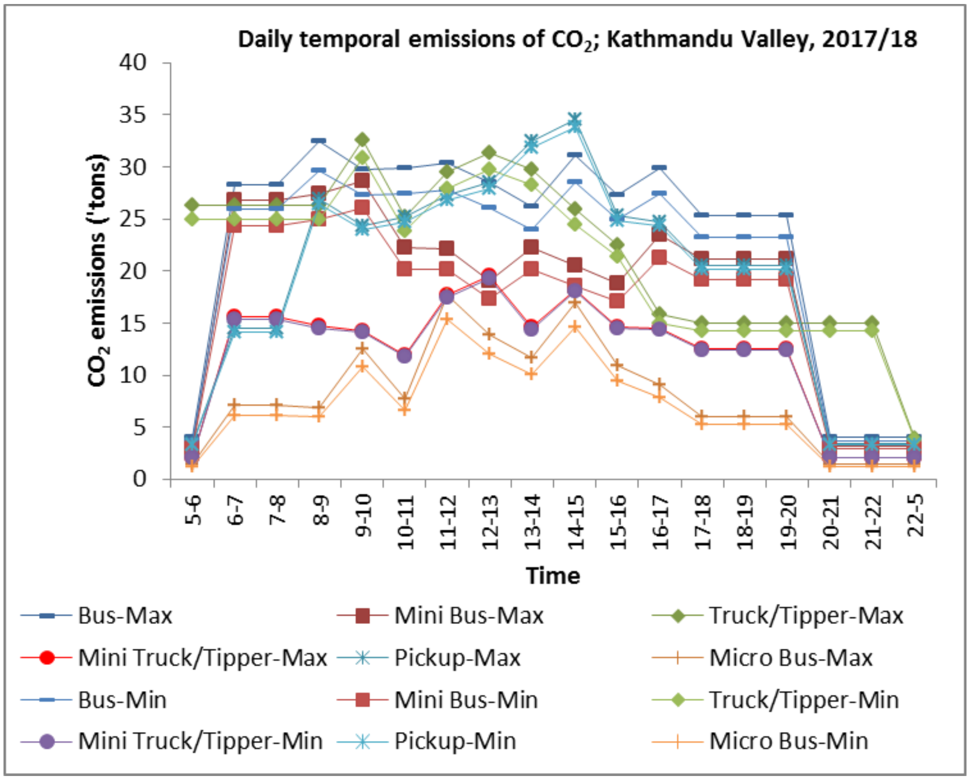

Figure 3.34: Daily temporal emissions of CO_2 in the Bagmati zone/ Kathmandu Valley

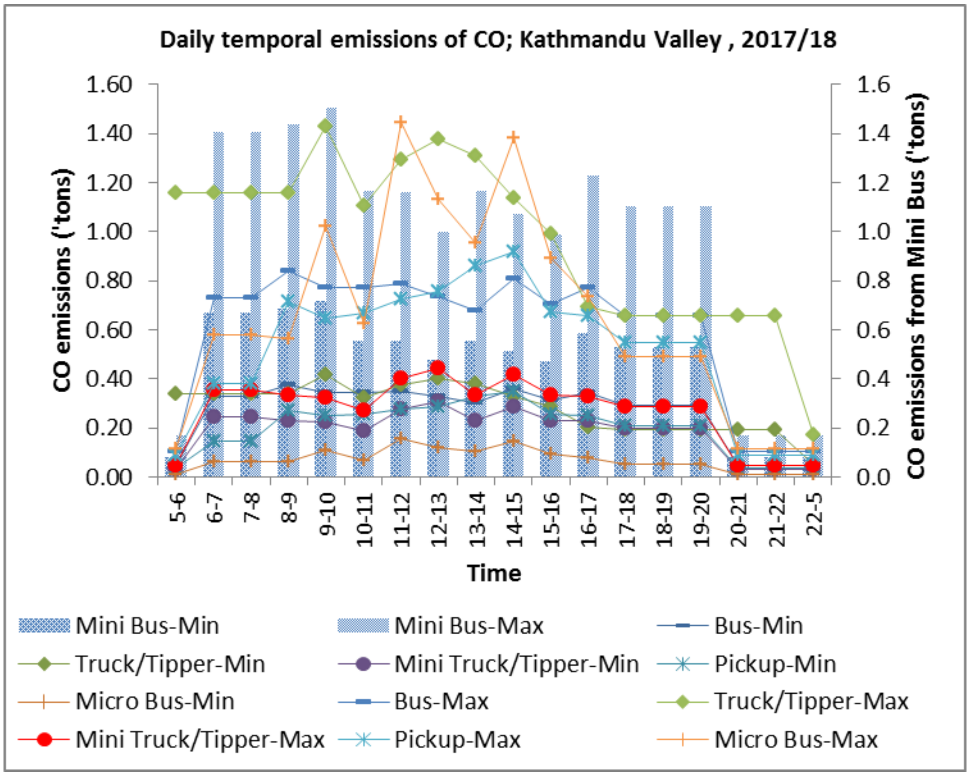

Figure 3.35: Daily temporal emissions of CO in the Bagmati zone/Kathmandu Valley

Similarly, the daily highest BC emission (35.8-178.3 kg) was at 11 am - 12 pm. The share of BC emissions at this time was by bus (24.7-31.5%), truck/tipper (12.5-35.4%), mini bus (5.7-25.7%), mini truck/tipper (6.2-7.9%), pickup (11.9–14.5%), and micro bus (9.6-14.4%). Similarly, the highest daily $PM_{2.5}$ emission (241.3-1449.2 kg) occurred between 2-3 pm (Das *et al.*, 2022). The share of $PM_{2.5}$ emissions at this time was by pickup (27.1-47.9%), mini truck/tipper (6.2-29.4%), bus (18.9-27.9%), truck/tipper (8.3-18.2%), mini bus (2.3-4.5%), and micro bus (4.1-5.1%) (Fig. 3.36–3.37).

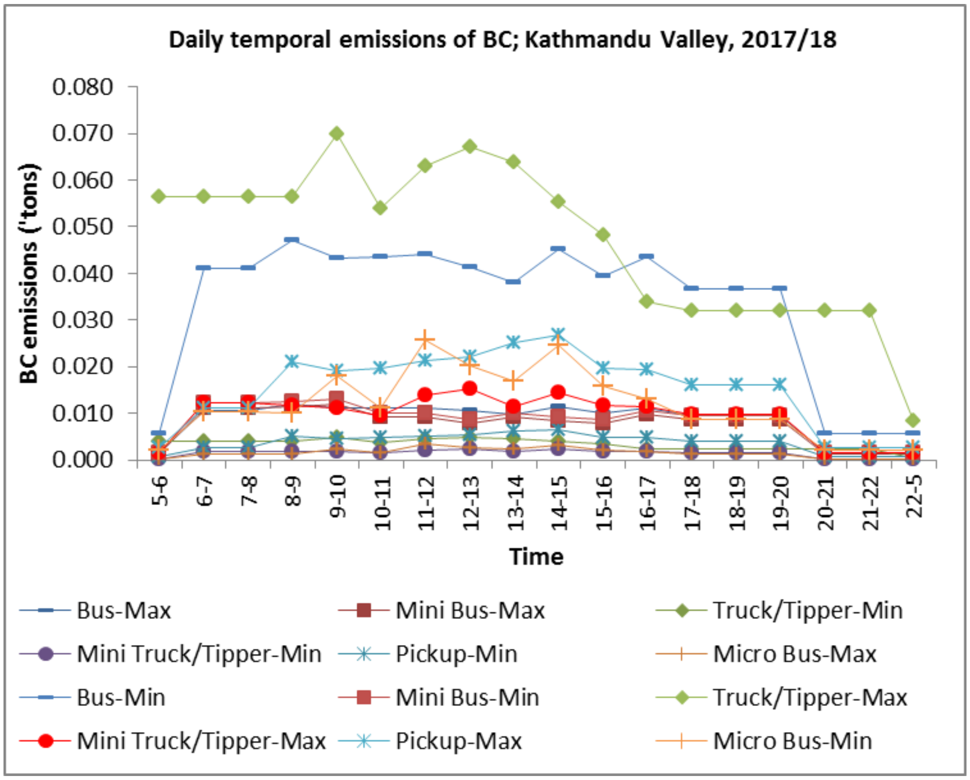

Figure 3.36: Daily temporal emissions of BC in the Bagmati zone/Kathmandu Valley

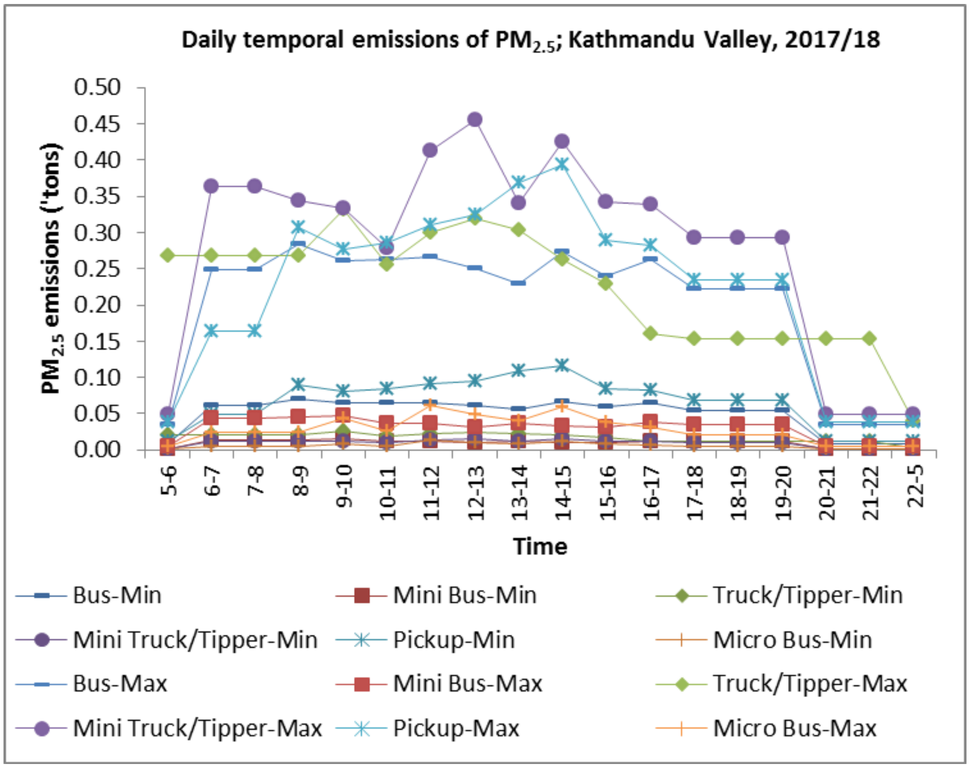

Figure 3.37: Daily temporal emissions of $PM_{2.5}$ in the Bagmati zone/Kathmandu Valley

As of past research, excessive traffic crowding, low fuel quality, and old and inadequately maintained road vehicles all contributed to a significant rise in emissions from the transport sector (Ale & Nagarkoti, 2003; Das *et al.*, 2018b; Mool *et al.*, 2020). Low vehicle speeds (< 2000 rpm) in crowded traffic situations lead to carbon deposits in the cylinders and injectors. This reduces engine performance even in brand-new vehicles and releases too much pollutant (Faiz *et al.*, 2006). This investigation found that 80% of samples surpassed the sulfur content of the BS III standard in a fuel quality test (350 mgL^{-1} as per NOC). BS II specifications were only met by two samples (compatible with EURO II) Moreover, lead (Pb) was also detected in the diesel fuel (Table 3.6).

Table 3.6: Fuel Quality Test Report

Diesel Pumps	**Location**	**Sample Date**	**Sulphur (%)**	**Lead (%)**
Private	Maharajgunj	03/04/2018	0.16	0.00025
Semi-government	Pulchowk	03/04/2018	0.13	0.00045
Government	Bhadrakali	30/03/2018	<0.05	<0.0005
Private	Kalanki	30/03/2018	<0.05	<0.0005
Private	Sundhara	01/04/2018	0.06	<0.0005
Private	Kalanki	30/03/2018	0.08	<0.0005
Government	Swoyambhu	30/03/2018	0.12	<0.0005
Private	Samakhusi	03/04/2018	0.19	0.002078
Private	Ekantakuna	03/04/2018	0.22	<0.0005
Private	Jadibuti	02/04/2018	0.08	0.001703

* Fuel test conducted at laboratory of Nepal Environmental and Scientific Services Pvt. Ltd., 2018.

All of this information indicates poor fuel quality in Nepal. In order to significantly cut emissions, it is crucial to look into cost-effective alternatives. The most essential intervention steps that could minimize air pollution are repairing and maintaining roads, improving fuel quality, and acquiring high-quality (> III euro grade) vehicles.

CHAPTER 4

SOUND SCIENTIFIC INFORMATION FOR REDUCING VEHICULAR AIR POLLUTION IN THE KATHMANDU VALLEY AND NEPAL

Abstract

One of the industries with the fastest growth is transport sector, which also contributes significantly to the increase in air pollution. Air pollution is currently a serious issue, owing to public health, climate change, and reduced visibility. Air pollution from the transportation sector is receiving a lot of attention in the media and policy arena right now. Although many industrialized nations have created regulations and methods for reducing vehicular emissions, it is still early in Nepal. Based on the reviewed literature, this study emphasizes the lessons learned from international policies, laws, and practices that have been applied to cut emissions from high emitting vehicles. In addition, present regulations and practices relating to vehicle emissions control in Nepal were investigated. This study summarizes the ideas, lessons learned, and research-based information offered in this document for sound policymaking in the transportation sector for improved air quality management in the Kathmandu Valley and Nepal.

Key words: Air quality; Emissions control; Policy; Transport sector, Vehicular emissions

4.1 Introduction

In Nepal, the transportation industry is increasing at the fastest rate, resulting in severe air pollution. Concerns have been raised over air quality, public health, climate change, and reduced visibility as a result of this. The media and policymakers are currently paying close attention to both the causes and the severe consequences. Although wealthy countries have enacted stringent rules to limit automotive emissions and improve air quality (Amann, 2017; Cai & Xie, 2013; Cui *et al.*, 2017; Holman *et al.*, 2015; Wu *et al.*, 2016), the situation in developing countries, such as Nepal, is far from ideal.

After recognizing the need for improved air quality, the National Environmental Policy (2019), National Transport Policy (2001), Vehicles and Transport Management Act (1992), National Climate Change Policy (2019), Environmental Conservation Act (2019), Nepal Vehicle Mass Emission Standard (2012), National Ambient Air Quality Standard (2012), Air Quality Management Action Plan of Kathmandu Valley (2019), and Nationally Determined Contribution (2020) were developed. These, on the other hand, do not provide an in-depth and well-defined framework for vehicle emissions control initiatives. In 1993, Nepal began measuring tail-pipe emissions in the Kathmandu Valley, and later in December 1999, the government enforced the Green Sticker. In an effort to reduce air pollution in the Kathmandu Valley, the government banned the importation of both new two-stroke vehicles and secondhand vehicles as well as the operating of very polluting diesel three-wheelers in 1999. As a result of this program, the use of electric vehicles (Safa tempo) for public transit has increased. From 2007 onward, the government began collecting NRs. 0.5 from every liter of fuel sold in the Kathmandu Valley as a pollution tax. In the year 2076 B.S., the Nepalese government changed the fuel tax, resulting in a threefold rise (i.e., NR 1.5 per liter) (Nepal Rajpatra, 2019.). However, the provision in the Financial Act 2002/03 directing the pollution tax to be deposited in the Environment Protection Fund has yet to be implemented, and the money collected has gone unused.

Air quality management remains one of the National Fifteenth Plan's primary objectives, as stated by Nepal's Constitution Assembly (amended in 2015 and 2020). Section 30 of constitution of Nepal highlights the need of a clean and healthy environment as a fundamental human right. Environmental conservation and management, as well as environmental friendly expansion, are critical phases in the nation's development that must be taken into account. The National Environmental Policy (2019) emphasizes clean air as a fundamental human right, as evidenced by objective 1. (i.e., prevention, control, and mitigation of emissions). The promotion of electric vehicles, hybrid vehicles, hydrogen-fueled vehicles, and other clean-fueled vehicles to be operated in Nepal is highlighted in No. 14 of the strategy under subheading 8.1. (No. 2) of subheading 8.5 envisions proper management for effective electric vehicles promotion, while No. 4 relates to pollution-reduction support for the renewable energy sector. To reduce pollution, heading 10 emphasizes the necessity to set base/guidelines for the transportation sector at the provincial level. The National Transport Policy (2001), includes a subheading 6.2 that calls for the prohibition of air-polluting vehicles, particularly in urban areas. It also emphasizes the prohibition on the import of vehicles older than five years. Clean energy sources such as gas, electricity, and solar power are also stressed for the operation of buses, trams, and other vehicles.

The criteria of vehicle pollution are highlighted in No. 23 (ga), paragraph 3 of the Vehicles and Transport Management Act (1992). The priority of vehicle registration is based on environmental pollution, vehicle/road pressure, road conditions, difficulties in vehicle operation, and other connected concerns, as stated in sub-divisions 1 and 2 of the Vehicles and Transport Management Act (act No. 45, 1992). Realizing the public welfare, DoTM has right to stop further partial or full registration of vehicles although articles 14 have provision of it (National Transport Policy, 2001). The National Climate Change Policy (2019) emphasizes the importance of a reliable transportation sector in achieving climate-resilient economic development. It takes into account the creation of emission reduction criteria for the transportation industry. Energy-efficient technology such as electric vehicles and the usage of electrical energy are recommended as solutions. It also emphasizes the phase-out of extremely polluting outdated transport vehicles in accordance with established requirements. The National Environmental Policy of 2019 also emphasizes it.

The second Nationally Determined Contribution (NDC) highlights the sales of electric vehicles (EV) to be 25% by 2025, which covers sales of private passenger vehicles and 2-wheelers, and 20% of all 4-wheeler public passenger vehicle sales. By 2030, the growth of electric vehicles (EVs) can cover up to 90% of all private passenger vehicle sales and 2-wheelers, and 60% of all 4-wheeler public passenger vehicle sales. It also emphasizes the deployment of 200 kilometers of electric rail network in the country by 2030 (NDC, 2020). If successful, this will result in a significant decline in emissions of pollutants from the transport sector.

4.2 Objectives

The general objective of this study was to highlight sound scientific information for policy review for reducing air pollution from transport sector in the Kathmandu Valley and Nepal.

The specific objectives were:

I. To review and highlight policies/legislation & practices, and identify policy gaps in the transport sector in Nepal.
II. To recommend policy options based on this study and international policy and practices.

4.3 Materials and Methods

The reports, policies, guidelines, and literature that had been published had been thoroughly evaluated and analyzed. The policies and practices that Nepal's government is currently pursuing to cut emissions were thoroughly investigated. The current national plan and efforts of the Nepalese government were also evaluated. Using a paper review, policy gaps in Nepal were found. Based on the policies and practices of other nations as well as scientific findings from this study, policy recommendations for curbing air pollution from transport sector for the Kathmandu Valley and Nepal are highlighted.

4.4 Results and Discussion

4.4.1 Environmental Guideline/Standard

The National Ambient Air Quality Standard (NAAQS) was first established in 2003 and was last revised in 2012. The Nepal Vehicle Mass Emission Standard (NVMES) was first implemented in 2000 and was last updated in 2012. All vehicles imported into Nepal, with the exception of heavy equipment vehicles, must now meet the EURO III emission standard. In order to increase fuel quality, Nepal Oil Corporation started supplying fuel that is compliant with the EURO VI standard in April 2020.

Nepal began monitoring tail-pipe emissions in the Kathmandu Valley in 1993, and the Green Sticker system has been in effect since December 1999. Transports' emission measurement was restricted to the Kathmandu Valley and only applied to 3-4 wheelers. The inspection and emission measurement, however, has not been effectively implemented.

The present emission standard of diesel vehicles for Nepal are depicted as below.

Table 4.1: Light duty diesel vehicles (< 2.5 tons)

Limit Values, Grams per Kilometer				
Type of Vehicle	Carbon Monoxide (CO)	Hydrocarbon (HC) and Nitrogen Oxide (HC+NOx)	Nitrogen Oxide (NOx)	PM
Passenger Car	0.64	0.56	0.5	0.05

Note: The driving cycle adopted by the vehicle manufacturing countries in accordance with the principles of EURO III driving cycle will be accepted.

Smoke Opacity Test	Free acceleration from idling to cut off speed. If Naturally aspirated: < or = 2.5 per meter If turbo charged: < or = 3.0 per meter

(Source: Ministry of Forest and Environment, 2067 B.S.)

Table 4.2: Light duty diesel vehicles (> 2.5 tons)

Limit Values, Grams per Kilometer				
Type of Vehicle	Mass of Carbon Monoxide (CO)	Combined Mass of Hydrocarbon (HC) and Nitrogen Oxide (HC+NOx)	Mass of Nitrogen Oxide (NOx)	Mass of Particulate Matters (PM)
LCV (RM= or < 1305 kg)	0.64	0.56	0.5	0.05
LCV (1305 > RM = or < 1760 kg)	0.8	0.72	0.65	0.07
LCV (RM > 1760 kg)	0.95	0.86	0.78	0.1

Note:

- The driving cycle adopted by the vehicle manufacturing countries in accordance with the principles of EURO III driving cycle will be accepted.
- RM signifies reference mass that represents unloaded vehicles with no driver and passengers but having full tank fuel with tools and spare tire adding another extra 100 kg weight or relative weight.

Smoke Opacity Test for diesel vehicle	Free acceleration from idling to cut off speed. If Naturally aspirated: < or = 2.5 per meter If turbo charged: < or = 3.0 per meter

(Source: Ministry of Forest and Environment, 2067 B.S.)

Table 4.3: Diesel vehicles (> 3.5 tons)

Limit Values, Grams per Kilometer				per meter
Carbon Monoxide (CO)	Mass of Hydrocarbon (HC)	Mass of Nitrogen Oxide (NOx)	Mass of Particulate Matters (PM)	Smoke
2.1	0.66	5	0.10 / 0.13**	0.8

Note:

- The driving cycle adopted by the vehicle manufacturing countries in accordance with the principles of EURO III driving cycle will be accepted.
- If the volume swept by the engine is lesser than 0.75 liter per cylinder and a rated power speed of more than 3,000 RPM, then the maximum m PM emission level is 0.13g/khz.

Smoke Opacity Test for diesel vehicle	Free acceleration from idling to cut off speed. If Naturally aspirated: < or = 2.5 per meter If turbo charged: < or = 3.0 per meter

(Source: Ministry of Forest and Environment, 2067 B.S.)

4.4.2 Air Quality Management Action Plan of Kathmandu Valley

Recognizing the grave apprehension of air pollution and its associated public health and climatic consequences, the Kathmandu Valley Air Quality Management Action Plan (2019) was developed, which covers transportation, solid waste, biomass open burning, and other related sectors (Table 4.4).

Table 4.4: The highlights of action plan that covers transport, solid waste and biomass open burning sectors.

Sectors	Activities	Timeline	Responsible authority	Supporting authority
Controlling emissions from transport sectors	Implementation of Euro IV grade vehicles	One year	Ministry of Forest and Environment	Department of Transport Management
	Development of in-used vehicle emission standard	One year	Ministry of Forest and Environment	Department of Transport Management and Department of Environment
	Issue of green sticker after passing from in-used vehicle emission standard	Regular	Department of Transport Management and Bagmati Province	Department of Environment and Traffic Division
	Enforcement of diesel particulate filter	Two years	Ministry of Forest and Environment	Department of Transport Management and Department of Environment
	Capacity building of vehicle workshops, increase in capacity/No., and its management	Two years	Department of Transport Management	Department of Environment
	Establishment of at least five vehicle emission tests center	1 year	Department of Transport Management and Bagmati Province	Department of Environment and Traffic Division
	Establishment of at least five vehicle fitness testing center	1 year	Department of Transport Management. Department of Environment and Traffic Division	
	Fuel quality guideline	2 years	Ministry of Forest and Environment, Department of Transport Management	
Promotion of electric vehicles	Development of charging stations at the bus terminals	1 year	Department of Transport Management, Ministry of Energy, Water Resources and Irrigation, Private Sectors	Nepal Electricity Authority, Ministry of Forest and Environment, Ministry of Physical Infrastructure & Transport
Ensuring fuel quality/adulteration check	Import based on prescribed guideline and ensuring fuel quality/adulteration check.	1 year	Nepal Bureau of Standards & Metrology, Department of Environment, Nepal Oil Corporation	Petrol pumps
	Biofuel blending guideline	1 year	Nepal Bureau of Standards &	Ministry of Forest and Environment,

			Metrology, Department of Environment, Nepal Oil Corporation, Traffic Division	Department of Environment
Inspection and monitoring of on-road vehicle emissions	At least five mobile inspection and monitoring of on-road vehicle emissions stations	1 year	Department of Transport Management. Department of Environment and Traffic Division	Bagmati Pradesh
Management of Industrial and waste	Impose fine and legal actions	Regular	Ministry of Forest and Environment, Bagmati Pradesh, Ministry of Social Development	Department of Public Health, Department of Industries
Household and agricultural waste management	Impose fine and legal actions	2 years	Ministry of Forest and Environment	Ministry of Forest and Environment, Department of Environment, Bagmati Pradesh
Fuel tax	Continuity of tax impose of fuel	Within one year and continuous	Ministry of Finance, Ministry of Forest and Environment	Department of Transport Management, Department of Roads, Department of Environment
	High tax on vehicle over 10 years	Within one year and continuous	Bagmati Pradesh, Department of Transport Management,	Ministry of Finance

Source: Air Quality Management Action Plan for Kathmandu Valley, 2019

4.4.3 Policy and intervention gaps

4.4.3.1 Vehicle emission standard

Vehicle emission standard (i.e., distance-based EFs) has been established by NVMES (2012). The higher permitted values (g/km) for passenger cars of light duty diesel vehicles (LDDV) (2.5 tons) are 0.64 CO, 0.56 (HC+NO_x), 0.5 NO_x, and 0.05 PM. Similarly, the permitted limit values (g/km) for LDDV (> 2.5 and < 3.5 tons) are 0.64-0.8 CO, 0.56-0.86 (HC+NO_x), 0.5-0.78 NO_x, and 0.05-0.1 PM. The upper allowed limit values (g/kWh) for diesel vehicles (> 3.5 tons) are 2.1 CO, 0.66 HC, 5.0 NO_x, and 0.10-0.13 PM (Table 25-27) (MoFE, 2067 B.S.). These tests, however, have yet to be applied in Nepal. Instead, for diesel vehicles, DoTM uses a smoke opacity test (SOT), which is confined to idling situations only. The SOT for diesel automobiles is 2.5 per meter (naturally aspirated) and 3.0 per meter (turbo charged). Diesel vehicles produce smoke at a density of 65 to 75 Hartridge Smoke Units (HSU) (Faiz *et al.*, 2006). Only the Kathmandu Valley has used-based vehicle emission testing, which is only applicable to three- and four-wheelers (Das *et al.*, 2018b).

Nepal's emission standard was compared to that of the rest of the world. For China's fifth stage, emission standards for HDDV are 4.0 g/kWh CO, 0.55 g/kWh HC, 2.8 g/kWh NO_x, and 0.03 g/kWh PM (ICCT, 2014). The highest allowed HDDV values in India (BS IV) are 1.5 g/kmhr CO, 0.96 g/kmhr HC, 3.5 g/kmhr NOx, and 0.02 g/kmhr PM (CPCB, 2017). Likewise, in Pakistan highest allowed value for HDDV for CO is 4 g/Kwh, HC (1.1 g/Kwh), NOx (7 g/Kwh) and PM (0.15 g/Kwh) (The Gazette of Pakistan, 2009). Similarly, the European Union's NO_x emission standards for Euro VI are 0.4 g/km (Vestreng *et al.*, 2009). In the USA, heavy-duty highway vehicles (Spark-Ignition) must meet emission standards of 0.195-0.230 g/mi NMHC, 0.2-0.4 g/mi NO_x, 0.02 g/mi PM, and 7.3-8.1 g/mi CO, depending on vehicle weight (USEPA, 2016). It is well clarified that the emission guidelines of Nepal are not in line with the developed countries (Das *et al.*, 2018b). Nepal's emission guidelines are, nevertheless, better than Pakistan's and poorer than Indian values.

4.4.3.2 Vehicle restriction to certain areas

Despite the fact that heavy duty vehicles are not permitted to use inside ring road during peak hour, the problem persists due to the ring road's proximity to the major market places. However, it has shown to be successful on a global scale (Das *et al.*, 2018b). After the execution of the Action Plan in 2013, HHDV was restricted to larger locations in China (e.g., Shanghai, Guangzhou, Shenzhen, and Hangzhou) (Wu *et al.*, 2016). In India, it has worked well. South Mumbai, a financial hub has banned high emitter vehicles to enter into the city areas from 7:00 a.m. to Midnight (Times of India, 25 January 2018). Countries in Europe with low emission zones and legislation, such as Austria, Denmark, Germany, the Netherlands, and Sweden, are similar to China (Holman *et al.*, 2015).

4.4.3.3 Inspection and maintenance

Vehicle emission tests are conducted by a sole organization, such as DoTM. Except for commercial vehicles, which are needed to be inspected every six months, testing is required once a year. After passing, the vehicle is given a green sticker, and if it fails, the vehicle is not authorized to operate in the Kathmandu Valley. According to one study, diesel vehicles can lower emissions by 33-40% with proper maintenance (Faiz *et al.*, 2006). According to Mool *et al.* (2020), timely servicing of diesel vehicles can significantly reduce emissions, reducing BC by 1.4 folds and $PM_{2.5}$ by 2.9 folds. In China, remote sensors are employed to inspect and measure diesel vehicles of high emitter on the road (Ministry of Environmental Protection, 2017). In Gothenburg, Sweden, vehicle inspection and maintenance techniques are also highly successful. In Nepal, there is currently no vehicle servicing policy, and also the inspection and emission testing system is ineffective (Das *et al.*, 2018b).

4.4.3.4 Fuel quality

In Nepal, the upper sulfur limit for diesel was 350 mg/kg (equiv. 350 ppm) (BS III), which has been in effect since 2010 (Das *et al.*, 2018b). In comparison to the global context, this value was significantly greater. Sulfur content was limited to 50 ppm (Euro IV) in China (Cai & Xie, 2013; Cui *et al.*, 2017), 50 ppm (BS IV) in India (Amann *et al.*, 2017; Guttikunda *et al.*, 2014), and 10 ppm (Euro VI) in Europe (Amann *et al.*, 2017; Guttikunda *et al.*, 2014). (ICCT, 2016). Nepal's imported fuel quality has been amended again, and is now in compliance with BS VI, as of April 2020 (NOC, 2022).

4.4.3.5 Higher engine technology

The Nepal's government has prohibited plying of extremely polluting three-wheeler diesel vehicles, two-stroke engines, and second-hand vehicles in the Kathmandu Valley since 1999 (Das *et al.*, 2018b). Going back to history, His Majesty Government of Nepal announced a plan to phase out 20 years old vehicles starting in 2000 (Faiz *et al.*, 2006).

Currently, all vehicles imported into Nepal, with the exception of HDDVs, must meet the EURO III emission standard. Engine technology has advanced significantly over the world. China has implemented Euro IV/V and electric vehicle standards (Cai & Xie, 2013; Cui *et al.*, 2017). In India, the Ministry of Road Transport and Highways has chosen to use Euro VI vehicles starting in 2020 (Amann, 2017). Euro VI has been in effect in Europe since 2014, with the goal of reducing emissions from the transportation sector (ICCT, 2016).

4.4.3.6 Subsidy, tax exemptions and scrappage of high-emitting vehicles

His Majesty's Government of Nepal had provisions to encourage EVs by waiving custom duty and sales taxes a decade ago. At present some steps are taken and the Nepal's government has considered reductions in excise tax, especially for low capacity of the EVs. It has, nevertheless, become a huge global concern. Subsidies and tax exemptions have been implemented by both local and federal governments in China to boost EV sales. In order to phase out vehicles with a yellow designation, a scrappage program was started using sufficient subsidies and incentives (Wu *et al.*, 2011). Similarly, Similarly, Europe is another example to provide enough subsidies/cut taxes to encourage EV sales. The excise tax decrease in Norway varies between 39% and 67%. In the Netherlands, France, and the United Kingdom, excise tax reductions range from 10% to 40% (European Commission, 2017). By 2040, Pakistan's National Electric Vehicle Policy seeks to have electric passenger vehicles and heavy-duty trucks account for 90% of sales, as well as electric two- and three-wheelers and buses (Udin, 2020). India anticipates deploying 5,595 electric buses across 63 cities as part of the Faster Adoption and Manufacturing of Hybrid and EV (FAME) initiative (SLOCAT, 2021). In its second NDC, Nepal has announced that by 2030, 90% of all passenger vehicles on the road should be electric. Vehicle sales (cars and two-wheelers) account for 60% of every four-wheeled passenger vehicle sold (SLOCAT, 2021).

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

A complete fine-resolution (1 km x 1 km) model-ready emissions inventory for open burning sectors (agricultural residue open burning and MSW open burning) was developed in this work for Nepal. Moreover, this study also identified the spatial and temporal characteristics of air pollutant emissions. For CROB, a yearly trend study was performed from 2003 to 2016, and for MSW open burning, from 2001 to 2016. In 2016/17, the mass of agriculture residue burned was 2,908 Gg (61-139%), accounting for 22% of the dry matter generation. From it, total emissions were estimated to be CO_2 4,140 Gg, CH_4 6.5 Gg (7-193%), $PM_{2.5}$ 24.5 Gg (30-170%), OC 8.6 Gg (38-162%), BC 2.2 Gg (-1-201%), CO 154 (4-196%), SO_2 1.2 (60-140%), NO_x 7 (54-146%), NMVOC 22.5 (8-192%) and NH_3 2.7 (3-197%), all in Gg/yr. The months of February to May accounted for more than 80% of all emissions, with April having the highest level of emissions.

According to the findings, CROB was less common in locations where farmers had more cattle per hectare. As a result, growing cattle and using dung by implementing household bio-gas plants by local farmers could be a viable mitigation option for reducing open burning, reducing harmful pollutants and GHGs, while also providing revenue. Another approach for mitigation is to use customized combine harvesters to chop crops and gather residue at the same time. Animal fodder, production of clean energy (e.g., bio-briquette/bio-char), and industrial raw materials can all benefit from the gathered leftovers (e.g., brick kilns, mushroom cultivation, paper production, etc.).

Similarly, the total quantity of MSW burned in Nepal in 2011 was estimated to be 89,000 tons (i.e., 240 tons/day), accounting for 4.5% of total trash generated that year. The SWM Act of 2011 does not include a section for open burning, which is currently one of the biggest problems. The revision of the SWM act of 2011 is critical for reducing air pollution from MSW open burning. The most effective MSW open burning control strategies and practices (e.g., improving waste segregation at the source, improving waste collection systems, banning MSW open burning at the community level, increasing penalty charges for use-throw practices as well as the effective implementation of apportioned fines by municipalities, extending more Kawadi (waste recycling) centers for waste recycling, systematic waste disposal practices, air curtain incineration practices, and raising awareness) are the important steps to reduce air pollution and its effect.

Furthermore, using experimentally based country-specific EFs, this study established a robust emission inventory from diesel vehicles for Kathmandu Valley and Nepal in between 1989/90 and 2017/18. The EFs were calculated through carbon balance technique for six diesel vehicle categories (bus, mini bus, truck/tipper, mini truck/tipper, pickup, and micro bus). Of total diesel vehicles studied (n = 29) at idle condition, the average EFs were calculated as CO_2 2600 (99-101%), CO 33.3 (44-156%), BC 0.6 (25-101%), and $PM_{2.5}$ 5.2 (0-235%) in unit of g L^{-1}. For on-road conditions (n = 5), the average EFs were estimated as 2,476 for CO_2 (90-110%), 97.3 for CO (0-232%), 1.7 for BC (46-110%), and 20.7 for $PM_{2.5}$ (0-255%), all in g L^{-1}. From 1989/90 to 2017/18, the average emission trend of CO_2, CO, BC, and $PM_{2.5}$ in the Bagmati zone/Kathmandu Valley and Nepal increased dramatically. Nationally, air pollutants emission from diesel vehicles were estimated to be CO_2 2,214 (90-110%) to 2,781(85-115%), CO 27.7 (42-158%) to 88.8 (0-232%), BC 0.51 (23-177%) to 3.55 (46-110%), and $PM_{2.5}$ 3.42 (0-236%) to 23.47 (0-255%) in the year 2017/18, all in Gg, of which 24.4-29.5%, 28.9-32.3%, 12.3-31.9%, and 21.8-42.5% was respectively for the Kathmandu Valley. The daily emissions for the year 2017/18 showed 138.3-147.2 tons of CO_2 and 241.3-1449.2 kg of $PM_{2.5}$ was peak at 2-3 p.m.; however, 36.8-178.3 kg of BC and 2.1-5.8 tons of CO was peak around 11 a.m.-12 p.m. in the valley. The major factors that affect variation in emissions could be vehicle types, age-wise vehicle statistics, vehicles' survival fraction, VKT, fuel utilization, efficiency, and choices of EFs.

Despite the fact that Nepal's constitution guarantee the people's basic right to live in a clean environment, air pollution levels continue to rise. National ambient guidelines and vehicle emissions standard must be strictly followed for better air quality management. Furthermore, in light of the findings of this study, existing policies and practices should be amended. This study recommends deeper insights of open burning sectors (e.g., agriculture and waste) in the National Environmental Policy (2019) and Climate Change Policy (2019). It also recommends for updating of national vehicle mass emission standards considering the research findings. National Transportation Policy (2001) should be amended to include and promote sustainable low-carbon transportation options (e.g., > Euro III grade engine, electric vehicles, high fuel quality with low sulfur and lead content, improved road conditions, timely repairing and maintenance of vehicles).

5.2 Recommendations

- The study's analysis of CROB takes into account CO_2 emissions from combustion in cropland, not net emissions. Numerous parameters must be taken into account in order to determine the net CO_2 emission; these were outside the scope of the current study but could be addressed in a future study.
- Growing cattle and using dung in residential bio-gas plants could be a potential mitigation options for lowering open burning from CROB, harmful pollutants and GHGs while also providing revenue for local farmers.
- To gather the chopped crop residue simultaneously with harvesting, the combine harvester should be adjusted. This leftover material can be used as animal feed, raw materials for industries, and for the production of renewable energy to clean up the air.
- Combine harvester should be modified to collect the chopped crop residue simultaneously while harvesting. This residue can be used as animal feed, renewable energy production and raw materials for industries to reduce air pollution.
- To reduce the uncertainties in MSW open burning, the site specific waste combustible fraction experiment, burning/oxidation efficiency (fraction), P_{frac}, and rural MSWGR are foremost steps for future research work.
- To reduce air pollution and its effects, more waste recycling centers should be established, as well as systematic waste disposal methods, air curtain incineration

practices, and efficient enforcement of penalties for open dumping and burning of MSW are crucial.

- This study further recommends in-depth study of country-specific EFs of pollutants from crop residue and MSW open burning to reduce uncertainties in the emissions.
- Deeper insights of open burning sectors (e.g., agriculture and waste) should be included in the National Environmental Policy (2019) and Climate Change Policy (2019).
- The foremost steps to reduce air pollutants from transport sector are repairing and maintenance of roads, timely repairing and maintenance of vehicles, and restrictions of construction vehicles, goods carrier & trucks to ply during official & peak hours especially in the Kathmandu Valley and other crowded cities of Nepal.
- This study also recommends revising vehicle emission standard (i.e., CO, BC, and $PM_{2.5}$) based on findings from this study followed by effective implementation.
- This study recommends revision of National Transport Policy (2001) by including and enhancing sustainable low carbon transport pathways (e.g. > BS III/Euro III vehicles, electric vehicles, hybrid vehicles, better fuel with low sulfur and lead content, clean & renewable energy fuel like hydrogen fuel, solar energy, subsidy, more tax exemptions, and scrappage of high emitter vehicles) followed by effective implementation.
- Further study on EFs by vehicle category (i.e., moving condition) by increasing the sample size is recommended.
- This study could not cover other diesel-powered tractors, jeeps/cars/vans, which could be the scope of future work.
- To reduce further uncertainty, a study on the average mileage of buses, minibuses, trucks, and mini trucks in the Nepalese context is recommended for future study.
- Referring to the data from this study, a separate research can be conducted (e.g., WRF modelling) to understand the air pollution behavior in the atmosphere.

CHAPTER 6

SUMMARY

This study established a complete high-resolution (1 km x 1 km) gridded model-ready emissions inventory from agriculture residue open burning for 2016/17 for the first time in the country. From 2003/04 to 2016/17, national air pollutant emissions from open burning of crop residue were estimated, with 2013/14 having the highest emissions. The total emissions for 2016/17 were estimated as CO_2 4,140 Gg, CH_4 6.5 Gg (7-193%), $PM_{2.5}$ 24.5 Gg (30-170%), OC 8.6 Gg (38-162%), BC 2.2 Gg (-1-201%), CO 154 (4-196%), SO_2 1.2 (60-140%), NO_x 7 (54-146%), NMVOC 22.5 (8-192%) and NH_3 2.7 (3-197%), all in Gg/yr. The months of February to May had the highest emissions (86.16% of total emissions), with April being the highest. Between July and September, however, emissions were almost negligible. In terms of per capita emissions from CROB, overall Nepal ranks 3 to 4 after Central Asia, China, and India. This study reveals that the Tarai region is exposed to more harmful pollutants due to CROB than the hills and mountains. The Tarai region has the largest BC emissions flux, followed by the hills and mountains. Residue burning was estimated to be negligible in districts where residue consumption demand exceeded generation need, which was connected closely with cattle raising too. According to this study, air quality in the Tarai region is more vulnerable to residue burning than in the hills and mountains. CROB is less common where farmers rear more cattle per hectare, according to this study. As a result, rearing cattle for additional income by local farmers could be a potential mitigating measure to prevent open burning. The deployment of modified combine harvesters, reaper harvesters, and happy seeders, as well as alternative uses of chopped residue for animal fodder, production of clean energy (e.g., bio-briquette/bio-char) and industrial raw materials (e.g., brick kilns, mushroom cultivation, paper production, etc.), are the most important steps to reduce CROB emissions.

Similarly, our work determined real-world emission parameters for estimating MSW open burning in Nepal. To understand the geographic characteristics of air pollutant emissions from MSW open burning, it is considered a comprehensive (1 km x 1 km) gridded emissions inventory. From 2001 to 2016, the national air pollutant emissions from MSW open burning were presented. In 2011, 89,000 tons of trash were burned (i.e., 240 tons/day), accounting for 4.5% of total waste generated that year. Over the last decade, there has been a 46% increase in MSW open burning. During 2011, a total of 8,790 tons of MSW were burned in urban areas (58 municipalities), while 80,000 tons were burned in rural regions (75 rural districts). According to this study, the annual trend of open burning in urban areas was much greater (350%) than in rural regions from 2001 to 2016. According to the findings, P_{frac} has a strong relationship with trash collection efficiency. The higher the P_{frac}, the lower the trash collecting efficiency, and vice versa. The experimental value of emission parameters can also be employed where waste composition and climatic conditions are similar to Kathmandu Valley municipalities. The methodologies used in this study can be used to estimate MSW open burning in other developing nations. In Nepal, the MSW open burning study emission inventory is quite primitive; therefore, additional research and development is required. This study's findings contribute to the development of a strong foundation for SWM policies that Nepal can implement in the near future. The most important steps to reduce this pollution and its effects are improving waste collection services, waste segregation at the source, banning MSW open burning, imposing a high penalty charge for open burning, establishing more Kawadi centers for waste recycling, systematic waste disposal practices (e.g., proper leachate control system, insects and pests control, mud cover, and tree plantation), and raising awareness.

This study also uses experimental-based EFs to predict diesel vehicle emissions in Nepal from 1989/90 to 2017/18. Bus, minibus, truck, tipper, mini tipper, pickup, and microbus were the vehicle categories chosen for this study. While the engine was idling and moving, Ratnoze1 and Microaeth were deployed to measure exhaust emissions. The carbon mass balance technique was applied to estimate the country-specific EFs of CO_2, CO, BC, and $PM_{2.5}$. Our findings suggest that the CO_2 EF was higher when the engine was idling than when it was driving. Moving vehicles, on the other hand, had greater CO, BC, and $PM_{2.5}$ levels than idling vehicles. The EFs were calculated as 2,559-2,605 for CO_2, 30.01-58.8 for CO, 1.03-1.59 for BC and 5.92-15.31 for $PM_{2.5}$ from buses, mini buses (2,271-2,574 for CO_2, 50-237 for CO, 0.6-2.2 for BC, and 1.6-5.4 for $PM_{2.5}$), mini truck/tipper (2,554-2,591 for CO_2, 39.6-62.5 for CO, 0.7-1.9 for BC, and 5.5-54.5 for $PM_{2.5}$), truck/tipper (2,564-2,602 for CO_2, 32.9-96.5 for CO, 0.4-2 for BC, and 3.5-16.3 for $PM_{2.5}$), pickup (2,596-2,610 for CO_2, 27.8- 35.8 for CO, 0.4-0.9 for BC, and 12.7-25.2 for $PM_{2.5}$), and micro bus (2,314-2,623 for CO_2, 19.7- 93.1 for CO, 0.4-1.3 for BC, and 2.1-7.3 for $PM_{2.5}$), in unit of g/L (Das *et al.*, 2022). Variation in results compared to other research could be related to factors such as vehicle types, age-wise vehicle statistics, vehicles' survival fraction, VKT, fuel utilization, efficiency, and choices of EFs.

On average CO_2, CO, BC, and $PM_{2.5}$ emissions from diesel vehicles have marked an increase. In the Bagmati zone/Kathmandu Valley, CO_2 emissions in the year 2017/18 were estimated as 637-679 Gg, 9.7-26.2 Gg of CO, 0.2-0.8 Gg of BC, and 1.0-6.5 Gg of $PM_{2.5}$, all higher than the 1989/90 value by 16.8-17.3, 5.1-6.6, 4.5-4.9, and 6.6-9.6 times, respectively. The CO_2 emissions in the year 2017/18 for Nepal were estimated as 2,214-2,781 Gg, 27.7-88.8 Gg of CO, 0.5-3.6 Gg of BC and 3.4-23.5 Gg of $PM_{2.5}$, higher by 13.1-16, 5.1-5.3, 3-4.9, and 5.9-7.5 folds, respectively, than the value for 1989/90. According to the findings, each pollutant varied in time. The Bagmati zone/Kathmandu Valley had the highest daily CO_2 emissions (138.3-147.2 tons) and $PM_{2.5}$ emissions (241.3-1449.2 kg) at 2-3 pm during the fiscal year 2017/18. On the other hand, the CO (2-5.8 tons) and BC (35.8-178.3 kg) were highest in between 11 a.m. and 12 p.m. Untimely vehicle repair and maintenance, irregularities in diesel filter change, a significant proportion of vehicles Euro III and lower, low fuel quality (Euro IV and lower), and diesel vehicle emission standards limited to the Smoke Opacity Test were all contributing to the rise in CO, BC, and $PM_{2.5}$ in Nepal. The best possible mitigation options are: revision of existing policy to include interventions such as vehicle inspection and maintenance; better fuel quality as lower grade fuel quality increases EFs and emissions; vehicle labeling (e.g., orange, yellow, green); restriction of high-emitting vehicles in urban areas; deployment of higher engine technology (> Euro III); government subsidy, more tax exemptions, and scrappage of high emitter vehicles; and research based vehicle emission standard.

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APPENDIXES

Appendix I: Uncertainties estimate of parameters of CROB through Monte Carlo method.

Uncertainties estimate of EFs.

	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
Lower (%)	80	14	17	89	53	43	9	76	18	12
Upper (%)	120	186	120	111	147	157	191	124	182	188
CV	0.10	0.44	0.43	0.06	0.24	0.29	0.46	0.12	0.42	0.45

Uncertainties estimate of residue burned.

	Residue burned (in % range)
Lower (%)	61
Upper (%)	139

Uncertainties estimate of emissions.

	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
Lower (%)	56	4	7	60	38	30	-1	54	8	3
Upper (%)	144	196	193	140	162	170	201	146	192	197

* 95% Confidence Interval converted to a percentage of the Mean

Appendix II: National dry matter generation (in Gg).

Year	Paddy	Maize	Millet	Wheat	Barley	Oil crops	Potato	Sugarcane	Jute	Pulses	Total
2003/04	5681.0	1272.1	272.0	1727.1	33.1	212.6	369.8	608.6	27.0	318.4	10521.7
2004/05	5469.5	1372.8	278.2	1795.8	31.7	227.2	391.2	627.3	25.9	325.6	10545.3
2005/06	5366.8	1387.5	279.3	1735.7	30.0	222.9	444.3	650.1	27.4	320.9	10465.0
2006/07	4693.1	1455.9	273.4	1886.3	30.5	217.1	437.2	686.3	26.9	329.3	10036.1
2007/08	5481.5	1502.9	278.4	1957.2	30.3	214.9	462.3	656.2	27.2	323.7	10934.7
2008/09	5767.7	1544.5	281.0	1673.1	25.1	216.8	545.4	621.6	28.3	306.5	11009.9
2009/10	5130.4	1484.1	287.5	1937.9	29.8	251.1	566.5	741.1	20.7	314.8	10764.0
2010/11	5686.9	1654.0	290.6	2173.5	32.6	281.9	564.3	717.6	23.1	380.2	11804.8
2011/12	6467.1	1743.5	302.5	2277.7	43.7	286.6	581.1	773.7	23.1	382.0	12880.9
2012/13	5743.2	1599.2	293.4	2343.4	39.9	286.4	599.4	773.5	24.8	428.6	12131.7
2013/14	6435.0	1826.6	291.9	2344.5	37.6	311.3	627.5	875.4	20.3	431.8	13201.9
2014/15	6105.5	1716.2	296.1	2459.6	40.3	335.4	581.9	889.8	20.1	431.8	12876.7
2015/16	5481.3	1482.2	287.2	1789.7	31.9	333.3	631.3	1147.5	18.6	460.2	11663.3
2016/17	6668.7	1840.1	294.4	2339.6	32.9	343.3	605.5	854.0	11.3	460.2	13450.0

Appendix III: Fuel consumption pattern (in Gg).

Year	Fuel energy	Roof thatching	Dry fodder
2003/04	1090.8	113.6	8816.9
2004/05	1117.1	109.4	8948.4
2005/06	1120.5	107.3	9056.7
2006/07	1149.7	93.9	9220.6
2007/08	1148.8	109.6	9363.9
2008/09	1174.8	115.4	9586.7
2009/10	1232.0	102.6	9739.3
2010/11	1257.0	113.7	9892.1
2011/12	1282.0	129.3	10029.1
2012/13	1307.0	114.9	10153.5
2013/14	1332.0	128.7	10074.4
2014/15	1392.7	122.1	10065.5
2015/16	1381.9	109.6	10118.7
2016/17	1406.9	133.4	10203.4

Appendix IV: Annual crop residue burning trend (in Gg).

Year	Residue to be burned	Burning Fraction (%)	M_1
2003/04	2638.5	25.08%	2279.7
2004/05	2487.6	23.59%	2149.3
2005/06	2315.7	22.13%	2000.8
2006/07	1924.7	19.18%	1663.0
2007/08	2548.5	23.31%	2202.0
2008/09	2512.0	22.82%	2170.4
2009/10	2103.1	19.54%	1817.2
2010/11	2695.9	22.84%	2329.3
2011/12	3438.9	26.70%	2971.3
2012/13	2785.0	22.96%	2406.3
2013/14	3532.8	26.76%	3052.4
2014/15	3211.5	24.94%	2774.8
2015/16	2463.1	21.12%	2128.2
2016/17	3365.3	25.02%	2907.7

Appendix V: National emissions trend from crop residue burning (in Gg).

Year	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
2003/04	3250	120	5.1	0.9	6.7	19.2	1.7	5.5	17.6	2.1
2004/05	3060	114	4.8	0.9	6.3	18.1	1.6	5.2	16.6	2.0
2005/06	2850	106	4.5	0.8	5.9	16.9	1.5	4.9	15.5	1.9
2006/07	2370	88	3.7	0.7	4.9	14.0	1.2	4.0	12.9	1.5
2007/08	3140	116	4.9	0.9	6.5	18.6	1.6	5.3	17.0	2.0
2008/09	3090	115	4.9	0.9	6.4	18.3	1.6	5.3	16.8	2.0
2009/10	2590	96	4.1	0.7	5.4	15.3	1.4	4.4	14.0	1.7
2010/11	3320	123	5.2	1.0	6.9	19.6	1.7	5.6	18.0	2.2
2011/12	4240	157	6.6	1.2	8.8	25.1	2.2	7.2	23.0	2.8
2012/13	3430	127	5.4	1.0	7.1	20.3	1.8	5.8	18.6	2.2
2013/14	4350	161	6.8	1.3	9.0	25.7	2.3	7.4	23.6	2.8
2014/15	3960	147	6.2	1.1	8.2	23.4	2.1	6.7	21.5	2.6
2015/16	3030	112	4.8	0.9	6.3	18.0	1.6	5.2	16.5	2.0
2016/17	4140	154	6.5	1.2	8.6	24.5	2.2	7.0	22.5	2.7
Avg.	3340	124	5.2	1.0	6.9	19.8	1.8	5.7	18.1	2.2

Appendix VI: Crop residue open burning emission inventory in the district level of Nepal in 2016/17 (in Tons).

District	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
Achham	19500	723	30.6	5.6	40.4	115	10.3	33.2	106	12.7
Baglung	10500	387	16.4	3.0	21.6	61.9	5.5	17.8	56.7	6.8
Bara	312000	11500	489	89.6	645	1840	164	530	1690	203
Bardiya	219000	8120	343	63.0	453	1300	115	372	1190	143
Bhaktapur	24300	900	38.1	7.0	50.2	144	12.8	41.3	132	15.8
Chitwan	82400	3050	129	23.7	171	488	43.3	140	447	53.7
Dang	120000	4460	189	34.6	249	713	63.3	205	653	78.5
Dhanusha	291000	10800	457	83.8	603	1720	153	496	1580	190
Gulmi	21000	777	32.9	6.0	43.4	124	11	35.7	114	13.7
Illam	16300	604	25.6	4.7	33.7	96.5	8.6	27.7	88.4	10.6
Jhapa	372000	13800	583	107	770	2200	195	632	2020	243
Kailali	288000	10700	452	82.8	596	1700	151	490	1560	188
Kalikot	63300	2350	99.3	18.2	131	375	33.3	108	344	41.3
Kanchanpur	192000	7130	302	55.3	398	1140	101	327	1040	126
Kapilbastu	150000	5550	235	43.1	310	887	78.7	255	813	97.7
Kathmandu	24800	921	39	7.1	51.4	147	13.1	42.3	135	16.2
Khotang	18000	667	28.2	5.2	37.3	107	9.5	30.6	97.6	11.7
Lalitpur	14300	529	22.4	4.1	29.6	84.5	7.5	24.3	77.5	9.3
Lamjung	27300	1010	42.9	7.9	56.6	162	14.4	46.5	148	17.8
Mahottari	58100	2150	91.1	16.7	120	344	30.5	98.8	315	37.9
Manang	22800	843	35.7	6.5	47.1	135	12	38.7	123	14.8
Morang	31700	1170	49.6	9.1	65.5	187	16.6	53.8	172	20.7
Nawalparasi	186000	6890	291	53.5	385	1100	97.7	316	1010	121
Parbat	11100	410	17.3	3.2	22.9	65.4	5.8	18.8	60	7.2
Parsa	280000	10400	440	80.6	580	1660	147	477	1520	183
Rasuwa	1240	46	1.9	0.4	2.6	7.3	0.7	2.1	6.7	0.8
Rautahat	173000	6400	271	49.6	357	1020	90.7	293	936	113
Rukum	1500	56	2.4	0.4	3.1	8.9	0.8	2.6	8.1	1
Rupandehi	299000	11100	468	85.9	618	1770	157	508	1620	195
Sarlahi	511000	18900	801	147	1060	3020	269	869	2770	333
Sindhupalchok	31800	1180	49.8	9.1	65.7	188	16.7	54	172	20.7
Siraha	146000	5430	230	42.1	303	866	76.9	249	794	95.5
Sunsari	26500	982	41.5	7.6	54.8	157	13.9	45	144	17.3
Surkhet	23100	856	36.2	6.6	47.8	137	12.1	39.3	125	15.1
Syangja	76100	2820	119	21.9	157	450	40	129	413	49.6
Total	4140000	154000	6500	1190	8580	24500	2180	7050	22500	2700

Note: Emissions are negligible in the other districts: Arghakhanchi, Baitadi, Bajhang, Bajura, Banke, Bhojpur, Dadeldhura, Dailekh, Darchula, Dhading, Dhankuta, Dolakha, Dolpa, Doti, Gorkha, Humla, Jajarkot, Jumla, Kaski, Kavre, Makwanpur, Mugu, Mustang, Myagdi, Nuwakot, Okhaldhunga, Palpa, Panchthar, Pyuthan, Ramechap, Rolpa, Salyan, Sankhuwashava, Saptari, Sindhuli, Solukhumbu, Tanahu, Taplejung, Terhathum, and Udayapur.

Appendix VII: Temporal estimates of emissions (in Gg).

Month	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
July, 2016	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
August, 2016	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
September, 2016	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
October, 2016	7.06	0.26	0.01	0.00	0.01	0.04	0.00	0.01	0.04	0.00
November, 2016	190.00	7.26	0.31	0.06	0.41	1.16	0.10	0.33	1.06	0.13
December, 2016	102.00	3.79	0.16	0.03	0.21	0.61	0.05	0.17	0.56	0.07
January, 2017	221.00	8.17	0.35	0.06	0.46	1.31	0.12	0.38	1.20	0.14
February, 2017	429.00	15.89	0.67	0.12	0.89	2.54	0.23	0.73	2.33	0.28
March, 2017	536.00	19.88	0.84	0.15	1.11	3.17	0.28	0.91	2.91	0.35
April, 2017	2250.00	83.44	3.53	0.65	4.66	13.32	1.18	3.83	12.21	1.47
May, 2017	355.00	13.14	0.56	0.10	0.73	2.10	0.19	0.60	1.92	0.23
June, 2017	47.64	1.77	0.07	0.01	0.10	0.28	0.03	0.08	0.26	0.03

Appendix VIII: Emissions (in tons) from MSW open burning from 2001-2016, Nepal.

	2001	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**
SO_2	51.0	51.2	51.6	52.0	52.6	53.3	54.3	55.5	56.9	58.5	60.4	62.6	65.1	67.9	71.1	74.8
CO	6469	6496	6545	6596	6671	6769	6891	7040	7217	7425	7665	7941	8257	8618	9027	9491
NMVOC	1145.7	1150.5	1159.0	1168.2	1181.4	1198.7	1220.4	1246.8	1278.1	1314.9	1357.4	1406.4	1462.3	1526.1	1598.6	1680.8
PM_{10}	611.0	613.6	618.1	623.0	630.1	639.3	650.9	665.0	681.7	701.3	724.0	750.1	779.9	813.9	852.6	896.4
CH_4	303.2	304.5	306.8	309.2	312.7	317.3	323.0	330.0	338.3	348.0	359.3	372.2	387.0	403.9	423.1	444.9
BC	252.1	253.1	255.0	257.0	259.9	263.7	268.5	274.3	281.2	289.3	298.6	309.4	321.7	335.7	351.7	369.8
OC	433.4	435.2	438.5	442.0	446.9	453.5	461.7	471.7	483.6	497.4	513.5	532.1	553.2	577.4	604.8	635.9
CO_2	122360	122869	123783	124762	126169	128020	130340	133157	136505	140426	144971	150199	156178	162991	170733	179514
NO_2	81.0	81.3	81.9	82.6	83.5	84.7	86.2	88.1	90.3	92.9	95.9	99.4	103.3	107.8	113.0	118.8
NO	116.1	116.6	117.4	118.4	119.7	121.5	123.7	126.3	129.5	133.2	137.6	142.5	148.2	154.6	162.0	170.3
NO_x	197.1	197.9	199.4	200.9	203.2	206.2	209.9	214.4	219.8	226.2	233.5	241.9	251.5	262.5	275.0	289.1
$PM_{2.5}$	562.9	565.3	569.5	574.0	580.4	589.0	599.6	612.6	628.0	646.0	666.9	691.0	718.5	749.8	785.5	825.9
NH_3	58.1	58.4	58.8	59.3	59.9	60.8	61.9	63.3	64.8	66.7	68.9	71.3	74.2	77.4	81.1	85.3

Appendix IX: Photographs of experiment on fraction of waste combustion


Measurment of sample mass of trash


Arrangement of sample of trash to be burnt


Sample of trash burning

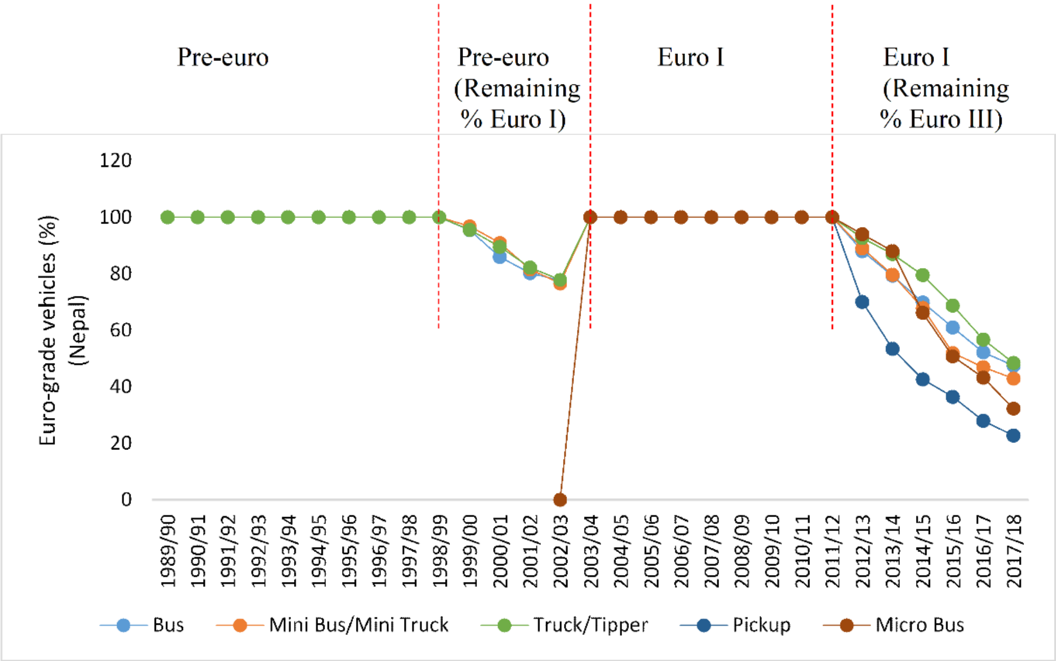

Sample of trash burning


Measurment of sample mass of trash and burning

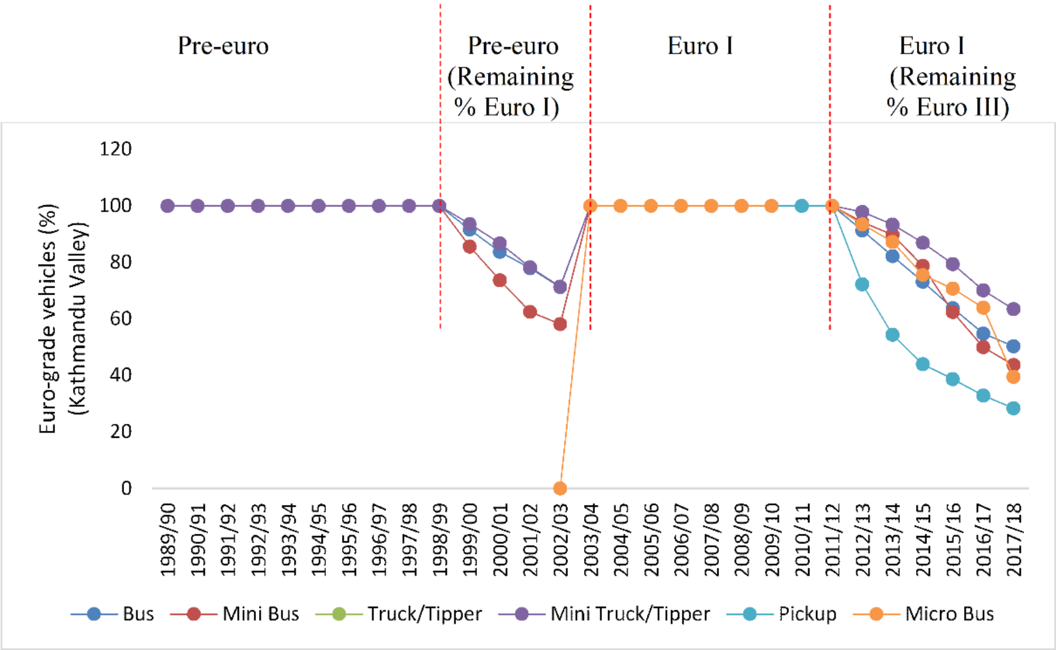

Segreggation of unburnt trash from burnt trash

Appendix X: Re-distribution of Euro-grade vehicles (Bagmati zone/Kathmandu Valley and Nepal).

(x)



(y)



Appendix XI: Uncertainties estimate of parameters of diesel vehicles through Monte Carlo method.

Uncertainty analysis through Monte Carlo Method.

Uncertainties estimate of diesel consumption through Monte Carlo method

	Fuel consumed (in % range)
Lower (%)	85
Upper (%)	115

Uncertainties estimate of EFs (idling condition, I) through Monte Carlo method

	EF CO_2-I	EF CO-I	EF BC-I	EF $PM_{2.5}$-I
Lower (%)	99	44	25	0
Upper (%)	101	156	101	235

Uncertainties estimate of EFs (moving condition, M) through Monte Carlo method

	EF CO_2-M	EF CO-M	EF BC-M	EF $PM_{2.5}$-M
Lower (%)	90	0	46	0
Upper (%)	110	232	110	255

Uncertainties estimate of emissions (idling condition) through Monte Carlo method

	CO_2-I	CO-I	BC-I	$PM_{2.5}$-I
Lower (%)	85	42	23	0
Upper (%)	115	158	177	236

Uncertainties estimate of emissions (moving condition) through Monte Carlo method

	CO_2-M	CO-M	BC-M	$PM_{2.5}$-M
Lower (%)	82	0	44	0
Upper (%)	118	234	156	256

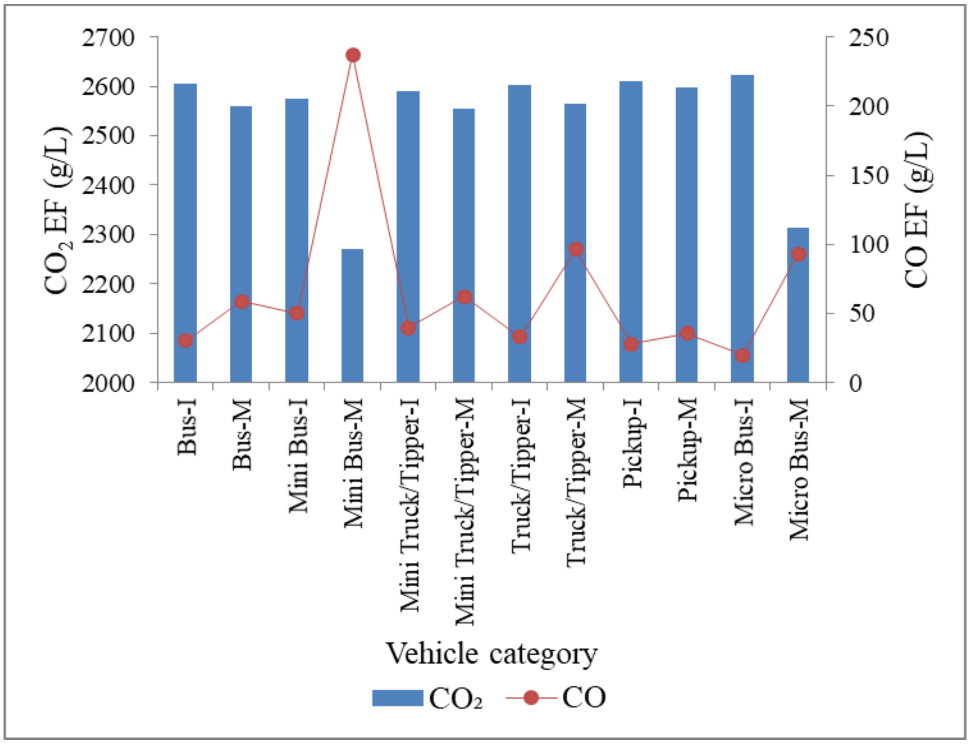
Appendix XII: Diesel consumption trend by vehicle category, Bagmati zone/Kathmandu Valley

Bus	Mini Bus	Truck/Tipper	Mini Truck/Tipper	Pickup	Micro Bus	Total (KL)	National sales, NOC (kL)	Bagmati zone consumption
3929	3932	4860	2670	0	0	15391	-	-
4364	4322	5825	3200	0	0	17712	-	-
4780	4689	6722	3693	0	0	19884	-	-
5221	4995	7731	4247	0	0	22194	-	-
6412	5145	10817	5942	0	0	28316	195689	14.5%
6929	5301	12095	6644	0	0	30970	226622	13.7%
7522	5443	13600	7471	0	0	34037	250500	13.6%
8197	6001	16408	9014	0	0	39620	257910	15.4%
9028	6617	17698	9723	0	0	43066	300604	14.3%
9763	7420	18845	10352	0	0	46380	315780	14.7%
10665	8488	20153	11071	0	0	50378	310569	16.2%
11664	10079	21743	11945	0	0	55431	326060	17.0%
12526	11884	24089	13233	0	0	61732	286233	21.6%
13722	12772	26413	14510	742	920	69078	299973	23.0%
14533	11946	27315	15006	1475	4508	74783	299730	25.0%
14902	13014	27905	15330	1475	6267	78893	315368	25.0%
18354	13305	30266	16627	1511	6448	86511	294329	29.4%
19849	14124	31072	17069	2551	6868	91532	306687	29.8%
21906	15417	33249	18265	4379	6958	100174	302706	33.1%
24689	20464	36306	19945	5799	7240	114443	446468	25.6%
30577	21734	44695	24554	7868	7655	137081	612505	22.4%
32148	22449	46338	25456	11099	7655	145145	655128	22.2%
33694	23168	46438	25511	14030	7974	150815	648513	23.3%
35600	24449	44349	24363	19442	8532	156735	716747	21.9%
38951	25494	45092	24771	25788	9143	169239	811100	20.9%
42904	28859	46685	25647	31959	10562	186617	901393	20.7%
48232	35549	47579	26137	36284	11285	205067	782451	26.2%
54523	43088	52067	28603	42681	12465	233428	1297066	18.0%
58010	47548	55740	30621	49395	20210	261525	1597551	16.4%

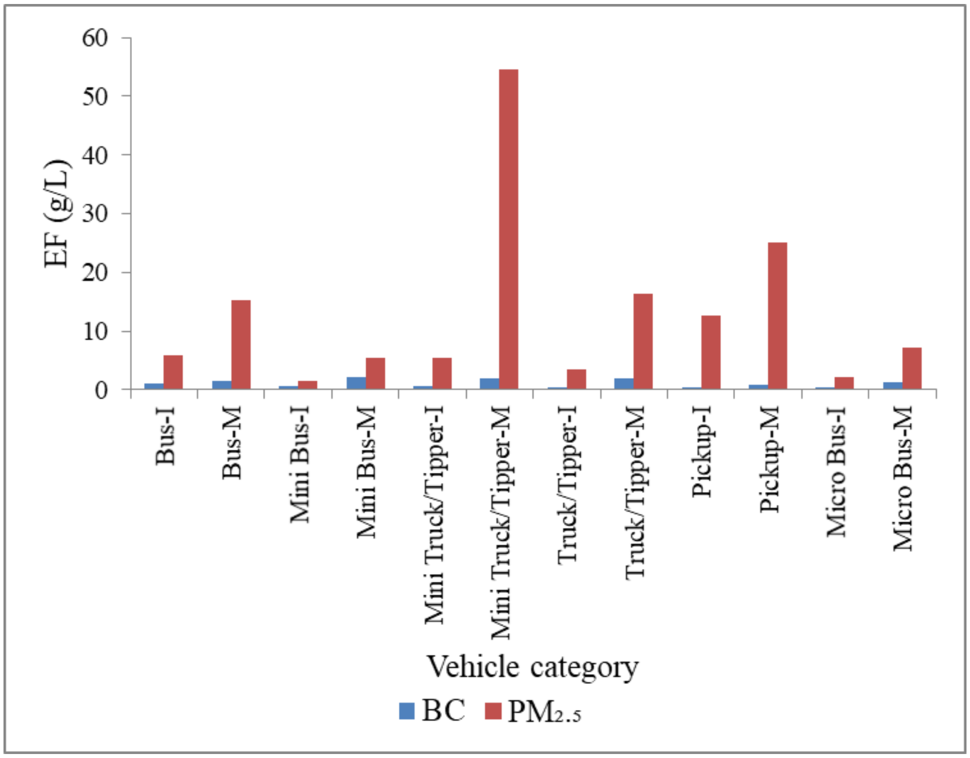
Appendix XIII: Diesel consumption trend by vehicle category, Nepal

Year	Bus	Mini Bus/Truck	Truck/Tipper	Pickup	Micro Bus	Total (KL)	National sales, NOC (kL)	National consumption
1989/90	20763	8466	37643	0	0	66872	-	-
1990/91	23154	10259	42451	0	0	75864	-	-
1991/92	25926	12125	51237	0	0	89288	-	-
1992/93	29089	12884	59833	0	0	101806	-	-
1993/94	35185	13380	69514	0	0	118079	195689	60.3%
1994/95	39622	13721	78906	0	0	132248	226622	58.4%
1995/96	42158	14057	85541	0	0	141756	250500	56.6%
1996/97	45332	14775	90568	0	0	150675	257910	58.4%
1997/98	50024	15308	98010	0	0	163343	300604	54.3%
1998/99	54576	15386	103649	0	0	173611	315780	55.0%
1999/00	57154	15887	108427	0	0	181468	310569	58.4%
2000/01	63433	16912	115755	0	0	196101	326060	60.1%
2001/02	67964	18861	126091	0	0	212916	286233	74.4%
2002/03	70219	20083	133047	1228	1233	225810	299973	75.3%
2003/04	70544	19630	135217	2238	5932	233561	299730	77.9%
2004/05	70979	19373	138008	2238	9037	239635	315368	76.0%
2005/06	75459	20668	144675	2314	9388	252503	294329	85.8%
2006/07	80126	22548	157094	3869	10121	273759	306687	89.3%
2007/08	84037	25959	171106	7225	10286	298614	302706	98.6%
2008/09	90161	26966	185017	9945	10966	323056	446468	72.4%
2009/10	108744	36925	242482	14119	11737	414008	612505	67.6%
2010/11	124231	40679	244576	20642	12348	442476	655128	67.5%
2011/12	131951	44719	243540	26942	13172	460324	648513	71.0%
2012/13	142886	49670	252318	38400	14012	497285	716747	69.4%
2013/14	152939	55121	258006	50377	14958	531402	811100	65.5%
2014/15	169907	64096	273963	63177	19913	591057	901393	65.6%
2015/16	189455	82350	309473	73870	25957	681104	782451	87.0%
2016/17	212645	90053	361613	96429	30427	791168	1297066	61.0%
2017/18	223606	98068	412100	118284	40708	892766	1597551	55.9%

Appendix XIV: EFs (CO_2 and CO) based on vehicle category



Appendix XV: EFs (BC and $PM_{2.5}$) based on vehicle category



Appendix XVI: Scaling Factors (SF) by Euro type

		CO_2 (SF)	CO (SF)	BC (SF)	$PM_{2.5}$ (SF)
Pre-euro to Euro III	Truck	1.00	2.96	5.74	2.51
Pre-euro to Euro III	Bus	1.00	1.32	5.74	2.51
Pre-euro to Euro III	LCV	0.95	4.29	5.74	2.51
Euro I to Euro III	Truck	0.94	1.29	3.75	1.99
Euro I to Euro III	Bus	0.87	1.29	3.75	1.99
Euro I to Euro III	LCV	1.16	1.29	3.75	2.90

Appendix XVII: Average Emission Factors (g/L) by Euro grade (Idling condition)

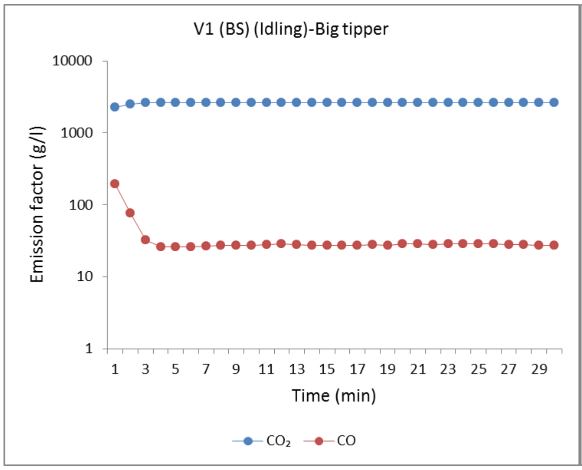
	Year	Vehicle grade	Fuel quality	CO_2	CO	BC	$PM_{2.5}$	Remarks
Bus	>2012	BS III (Euro III)	Euro IV	2612	26.4	0.3	3.6	Measured value
	1999-2011	BS I (Euro I)	Euro IV	2598	33.4	1.6	7.6	Measured value
	<1998	Pre-euro	Euro IV	2622	34.9	1.8	9.1	Estimated value
Mini bus/truck /tipper	>2012	BS III (Euro III)	Euro IV	2605	30.5	0.5	2.2	Measured value
	1999-2011	BS I (Euro I)	Euro IV	2587	42.0	0.4	2.0	Measured value
	<1998	Pre-euro	Euro IV	2487	131.0	2.6	5.5	Estimated value
Truck/tipper	>2012	BS III (Euro III)	Euro IV	2602	32.1	0.5	3.6	Measured value
	1999-2011	BS I (Euro I)	Euro IV	2600	33.8	0.3	1.1	Measured value
	<1998	Pre-euro	Euro IV	2594	95.0	2.9	9.0	Estimated value
Pickup	>2012	BS III (Euro III)	Euro IV	2614	24.7	0.6	3.5	Measured value
	1999-2011	BS I (Euro I)	Euro IV	2605	30.9	0.3	21.8	Measured value
	<1998	*Vehicle not existed*						
Micro bus	>2012	BS III (Euro III)	Euro IV	2612	26.3	0.6	1.7	Measured value
	1999-2011	BS I (Euro I)	Euro IV	2626	17.4	0.3	2.2	Measured value
	<1998	*Vehicle not existed*						

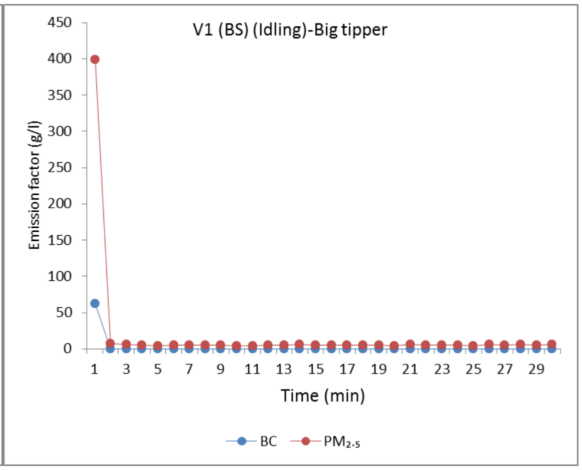
** BS (Bharat Stage)*

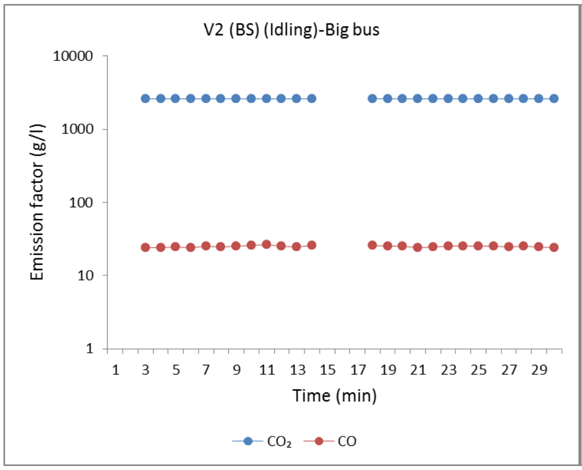
Appendix XVIII: Average Emission Factors (g/L) by Euro grade (Moving condition)

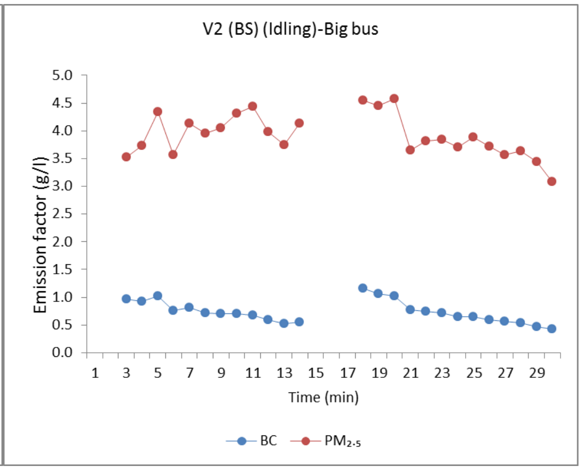
	Year	Vehicle grade	Fuel quality	CO_2	CO	BC	$PM_{2.5}$	Remarks
Bus	>2012	BS III (Euro III)	Euro IV	2559	58.8	1.6	15.3	Measured value
	1999-2011	BS I (Euro I)	-	2214	75.7	6.0	30.4	Estimated value
	<1998	Pre-euro	-	2569	77.7	9.1	38.5	Estimated value
Mini bus/truck/tipper	>2012	BS III (Euro III)	Euro IV	2460	120.7	2.0	38.2	Measured value
	1999-2011	BS I (Euro I)	-	2843	155.3	7.5	62.3	Estimated value
	<1998	Pre-euro	-	2348	517.7	11.5	95.9	Estimated value
Truck/tipper	>2012	BS III (Euro III)	Euro IV	2564	96.5	2.0	16.3	Estimated value
	1999-2011	BS I (Euro I)	-	2404	124.2	7.6	32.3	Estimated value
	<1998	Pre-euro	-	2557	285.9	11.6	40.9	Estimated value
Pickup	>2012	BS III (Euro III)	Euro IV	2596	35.8	0.9	25.2	Measured value
	1999-2011	BS I (Euro I)	-	2478	153.6	5.0	41.2	Estimated value
	<1998	*Vehicle not existed*						
Micro bus	>2012	BS III (Euro III)	Euro IV	2314	93.1	1.3	7.3	Estimated value
	1999-2011	BS I (Euro I)	-	2208	399.6	7.6	11.9	Estimated value
	<1998	*Vehicle not existed*						

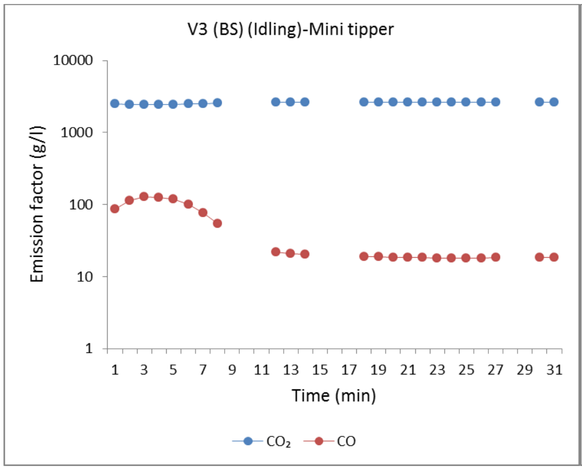
Appendix XIX: Findings of EFs by vehicle category

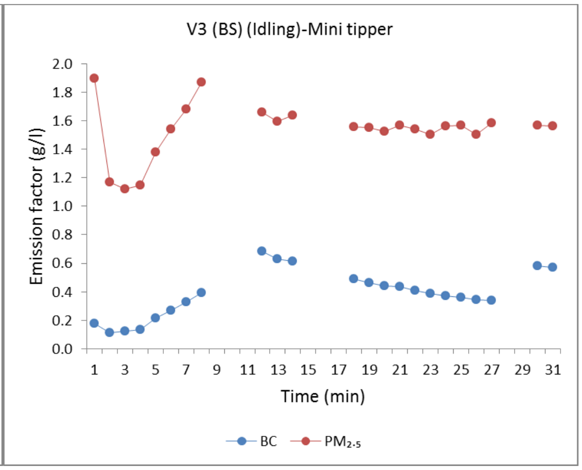


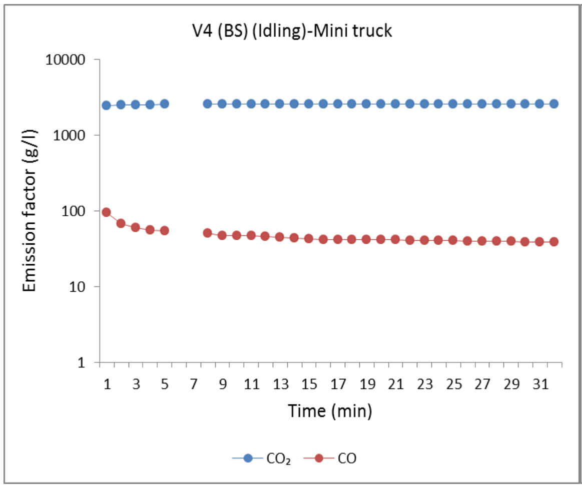


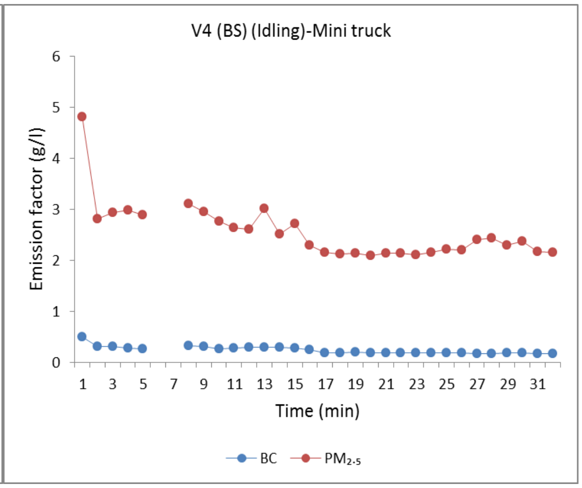


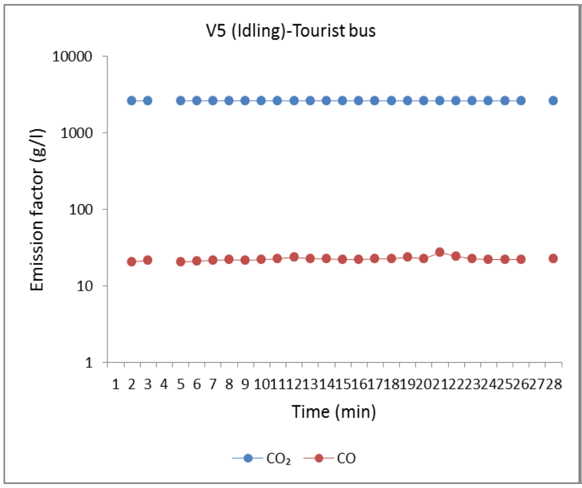


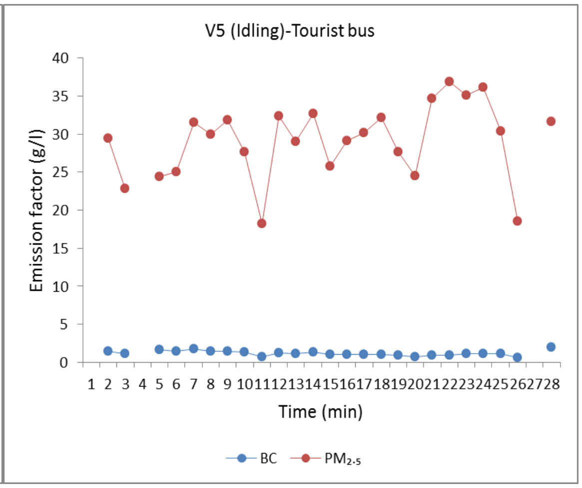


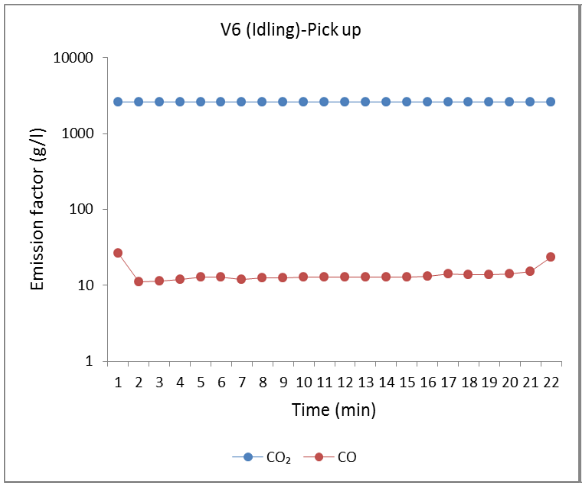


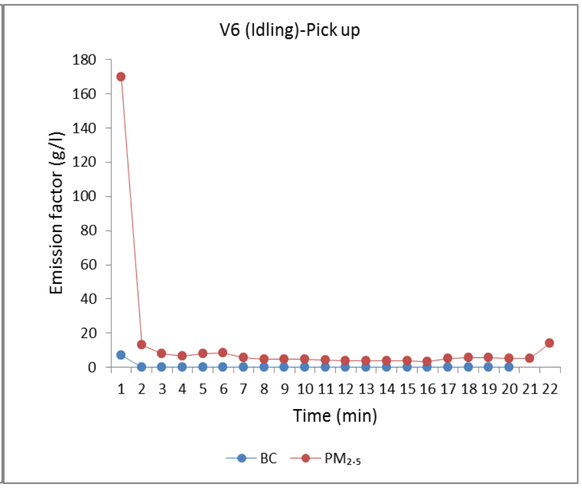

V4 (BS) (Idling)-Mini truck
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
CO₂
CO

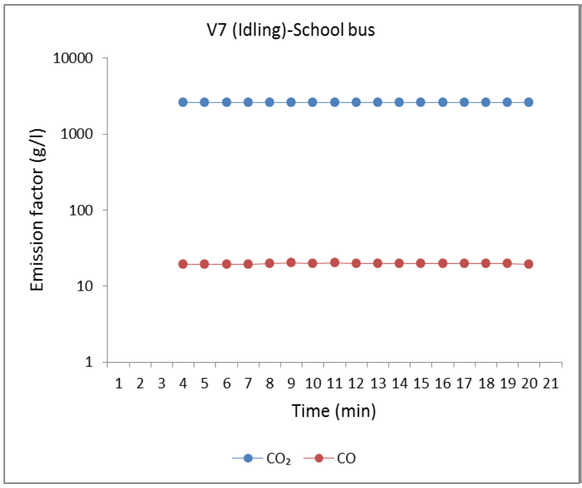

V4 (BS) (Idling)-Mini truck
Emission factor (g/l)
6
5
4
3
2
1
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
BC
PM₂.₅

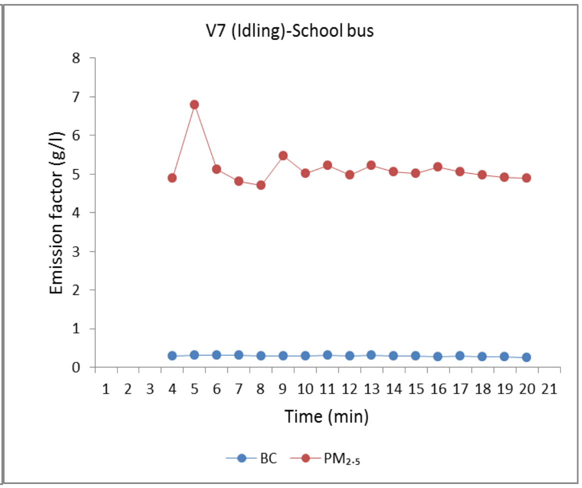

V5 (Idling)-Tourist bus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28
Time (min)
CO₂
CO

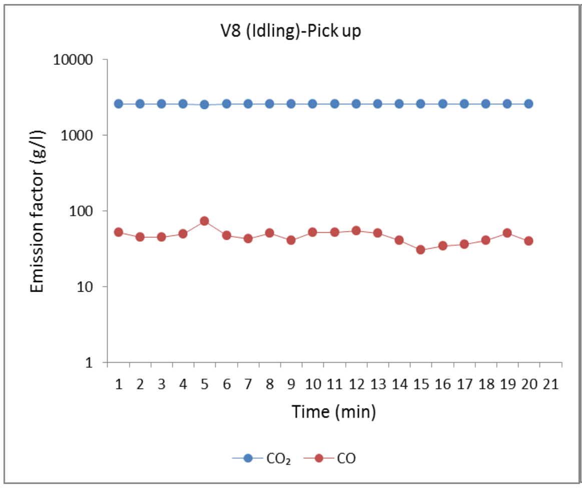

V5 (Idling)-Tourist bus
Emission factor (g/l)
40
35
30
25
20
15
10
5
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28
Time (min)
BC
PM₂.₅

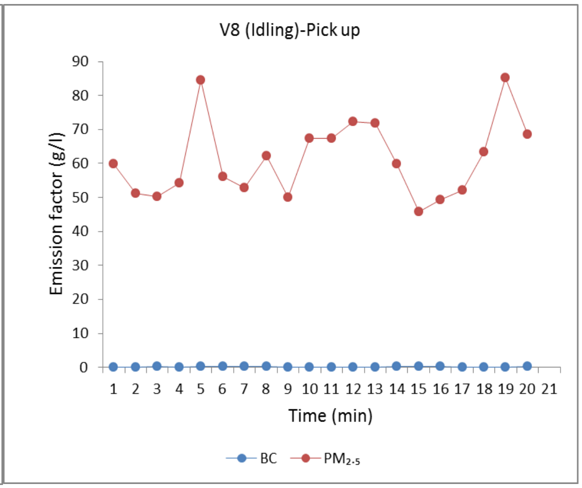

V6 (Idling)-Pick up
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22
Time (min)
CO₂
CO

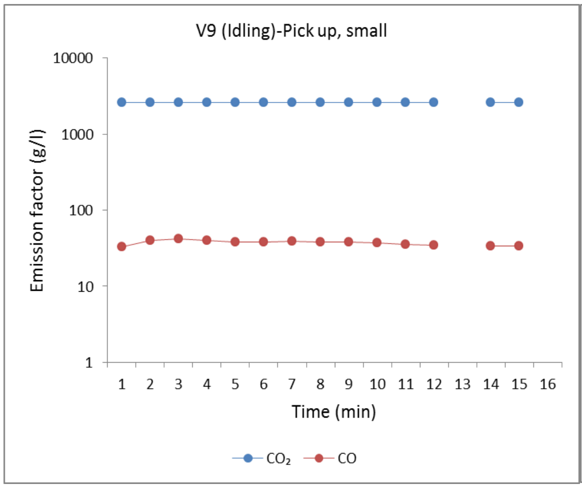

V6 (Idling)-Pick up
Emission factor (g/l)
180
160
140
120
100
80
60
40
20
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22
Time (min)
BC
PM₂.₅

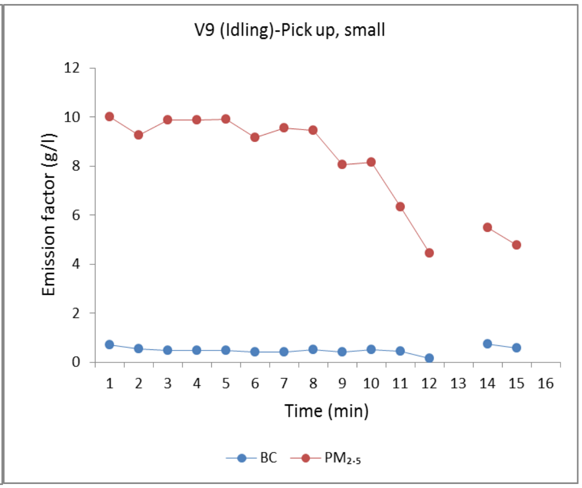

V7 (Idling)-School bus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
CO₂
CO

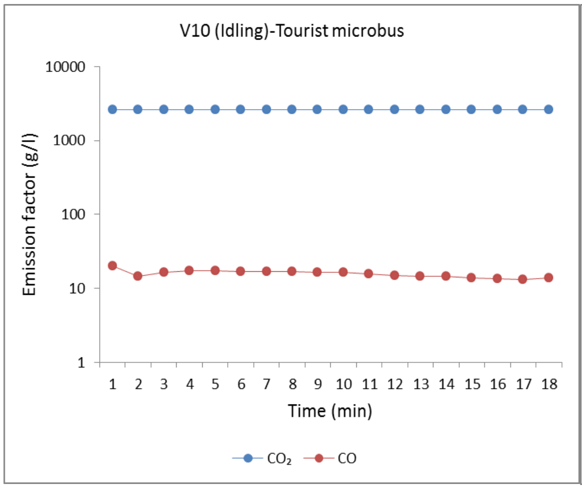

V7 (Idling)-School bus
Emission factor (g/l)
8
7
6
5
4
3
2
1
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
BC
PM₂.₅

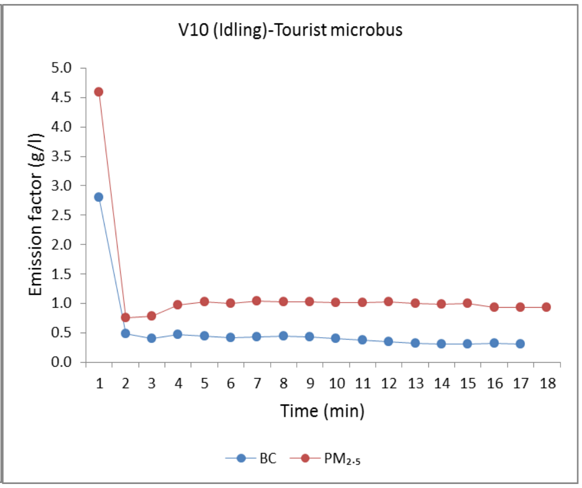

V8 (Idling)-Pick up
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
CO₂
CO

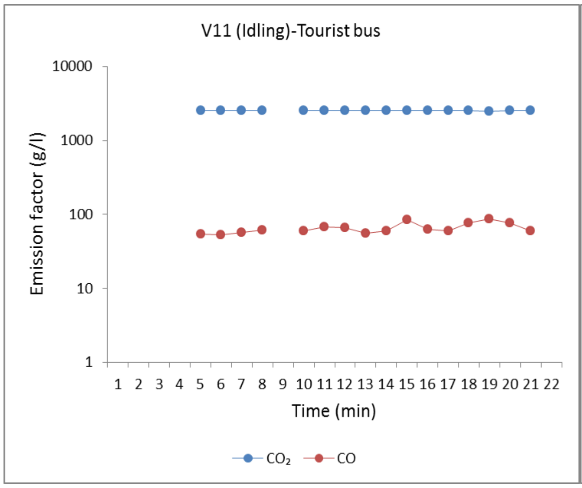

V8 (Idling)-Pick up
Emission factor (g/l)
90
80
70
60
50
40
30
20
10
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
BC
PM₂.₅

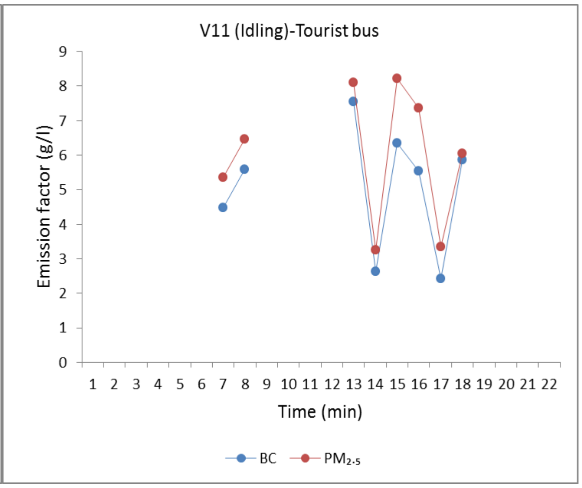

V9 (Idling)-Pick up, small
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16
Time (min)
CO₂
CO

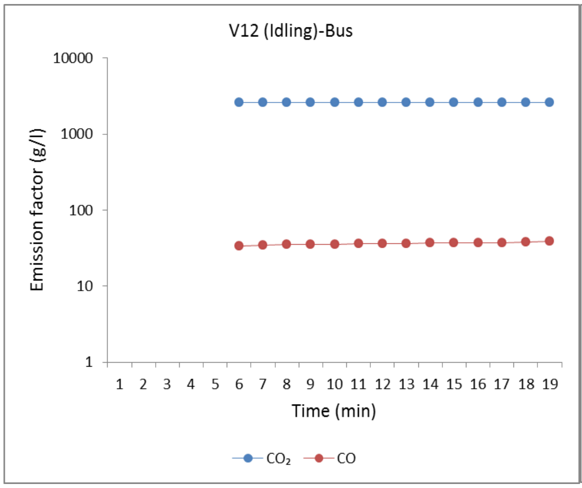

V9 (Idling)-Pick up, small
Emission factor (g/l)
12
10
8
6
4
2
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16
Time (min)
BC
PM₂.₅

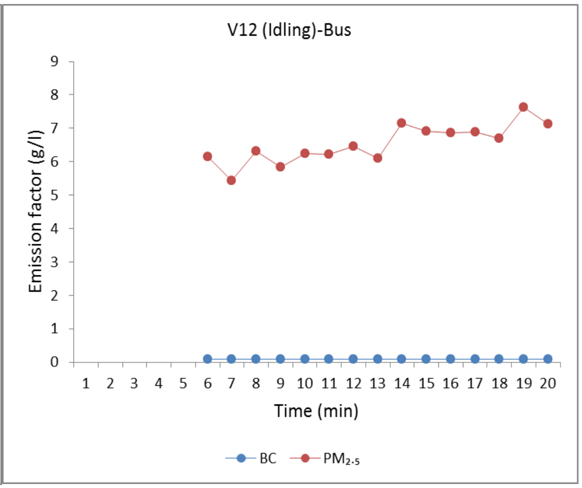

V10 (Idling)-Tourist microbus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18
Time (min)
CO₂
CO

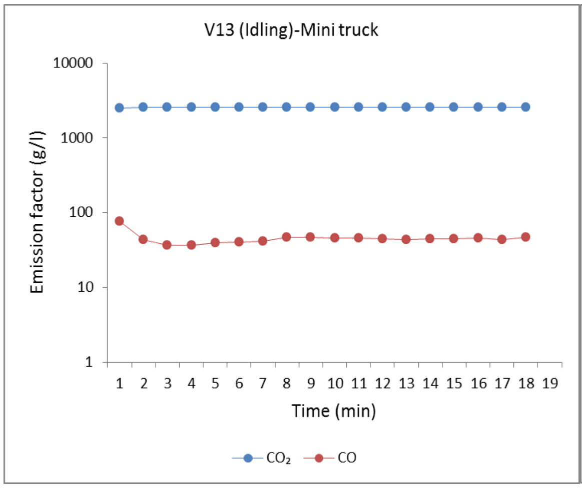

V10 (Idling)-Tourist microbus
Emission factor (g/l)
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18
Time (min)
BC
PM2.5

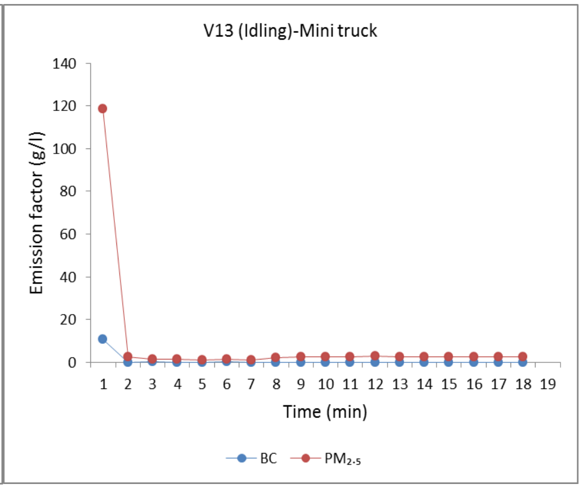

V11 (Idling)-Tourist bus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22
Time (min)
CO₂
CO

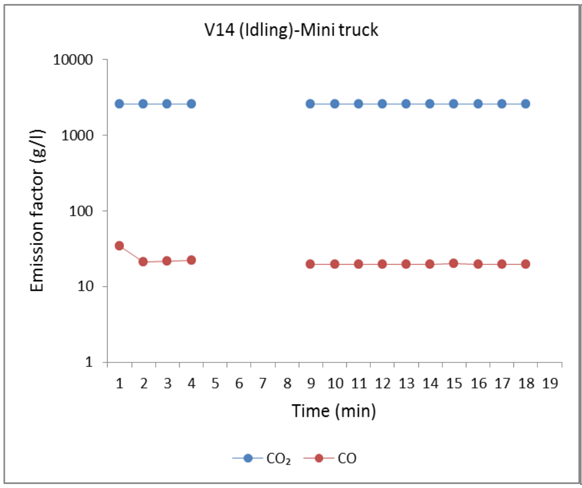

V11 (Idling)-Tourist bus
Emission factor (g/l)
9
8
7
6
5
4
3
2
1
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22
Time (min)
BC
PM2.5

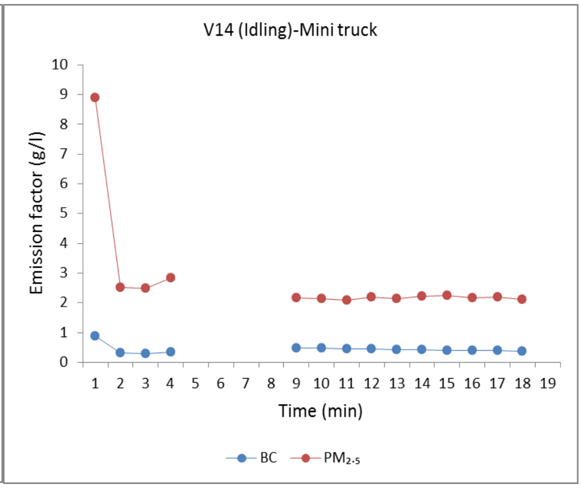

V12 (Idling)-Bus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
CO₂
CO

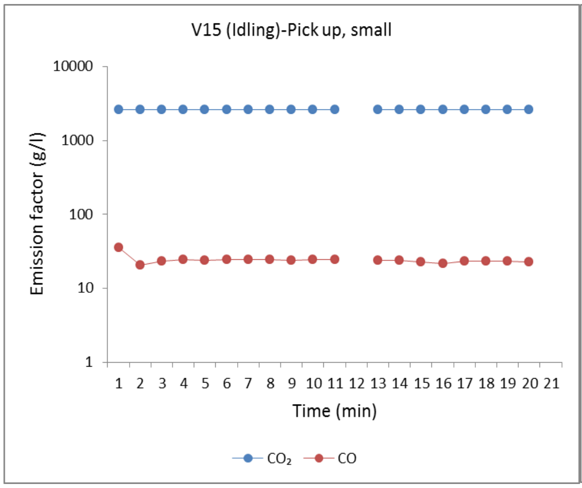

V12 (Idling)-Bus
Emission factor (g/l)
9
8
7
6
5
4
3
2
1
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20
Time (min)
BC
PM2.5

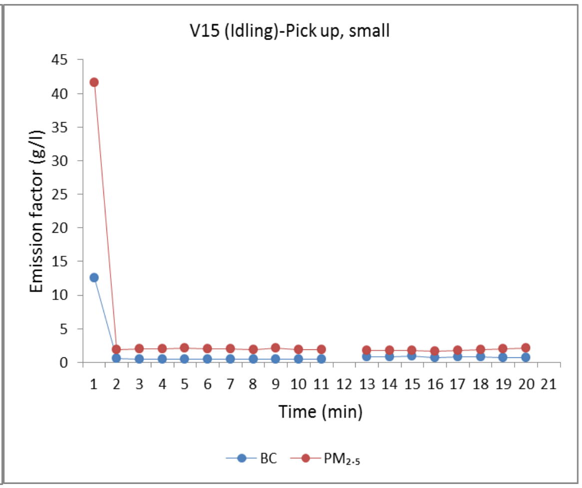

V13 (Idling)-Mini truck
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
CO₂
CO

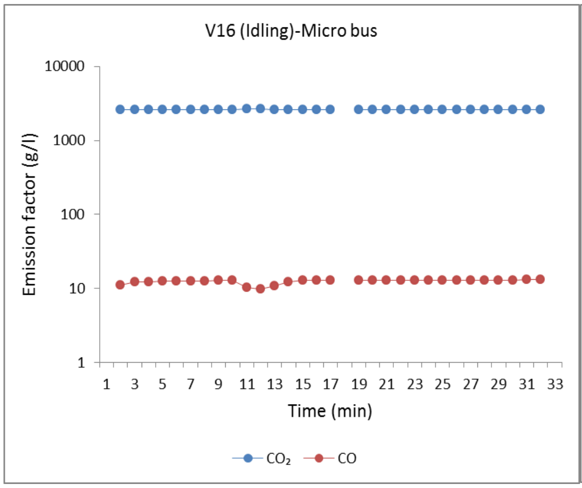

V13 (Idling)-Mini truck
Emission factor (g/l)
140
120
100
80
60
40
20
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
BC
PM₂.₅

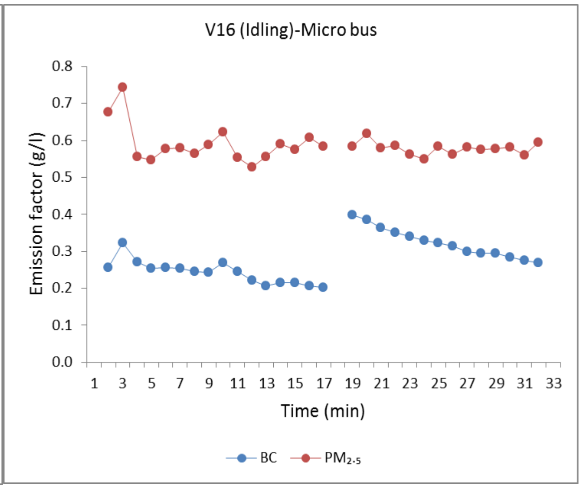

V14 (Idling)-Mini truck
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
CO₂
CO

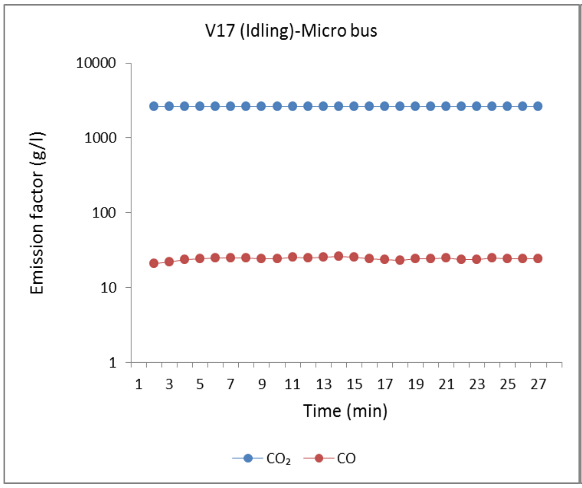

V14 (Idling)-Mini truck
Emission factor (g/l)
10
9
8
7
6
5
4
3
2
1
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
BC
PM₂.₅

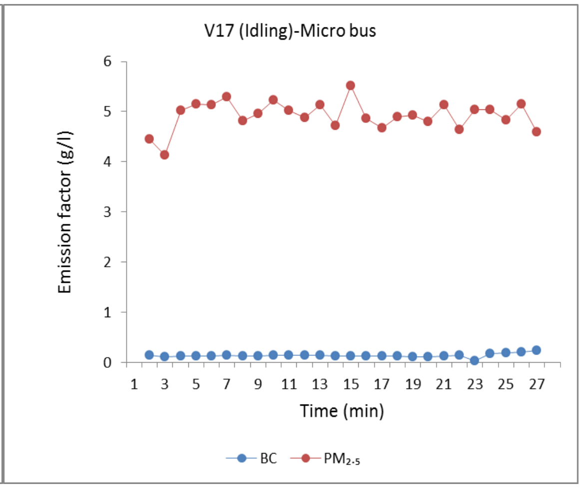

V15 (Idling)-Pick up, small
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
CO₂
CO

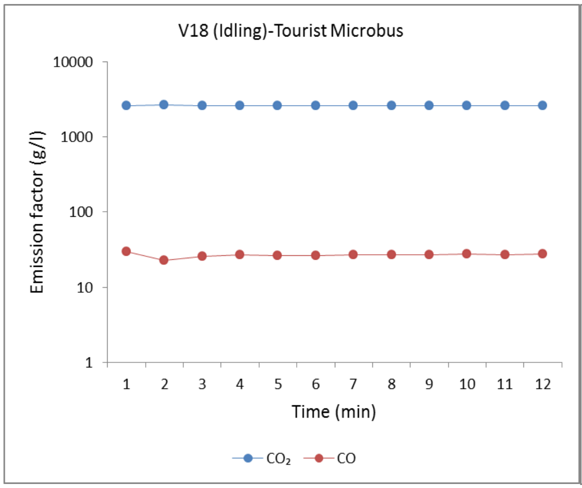

V15 (Idling)-Pick up, small
Emission factor (g/l)
45
40
35
30
25
20
15
10
5
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21
Time (min)
BC
PM₂.₅

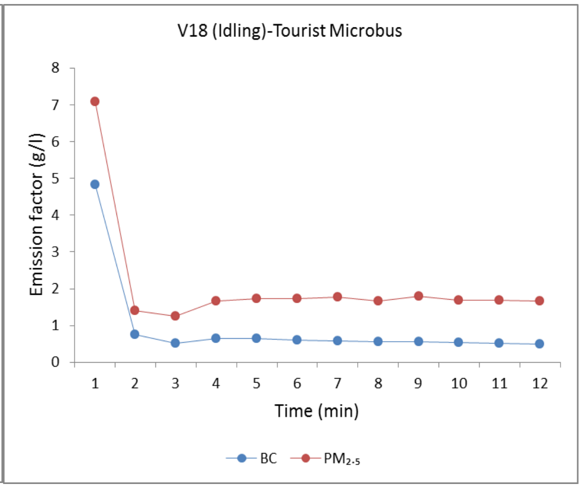

V16 (Idling)-Micro bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33
Time (min)
CO₂
CO

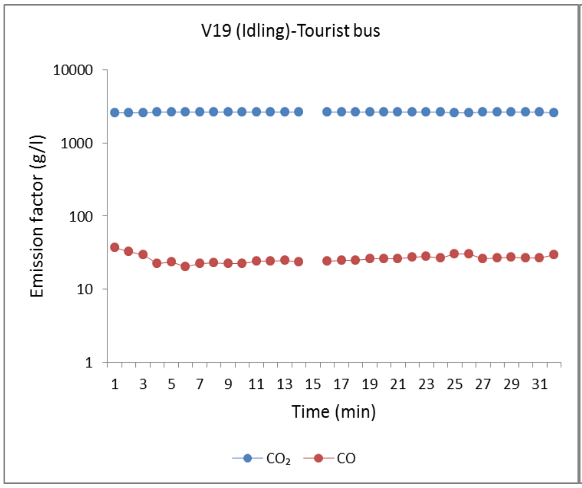

V16 (Idling)-Micro bus
Emission factor (g/l)
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33
Time (min)
BC
PM₂.₅

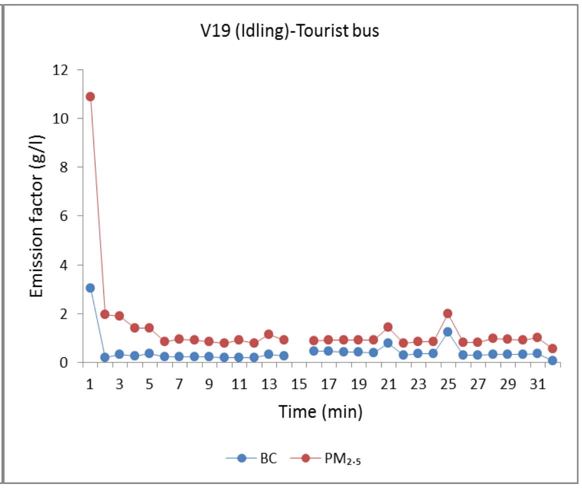

V17 (Idling)-Micro bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27
Time (min)
CO₂
CO

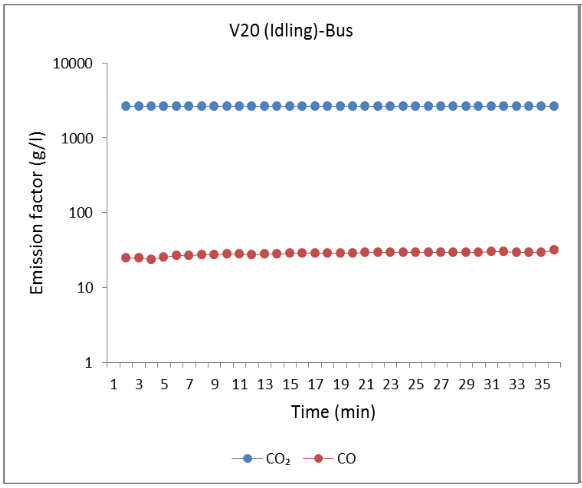

V17 (Idling)-Micro bus
Emission factor (g/l)
6
5
4
3
2
1
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27
Time (min)
BC
PM₂.₅

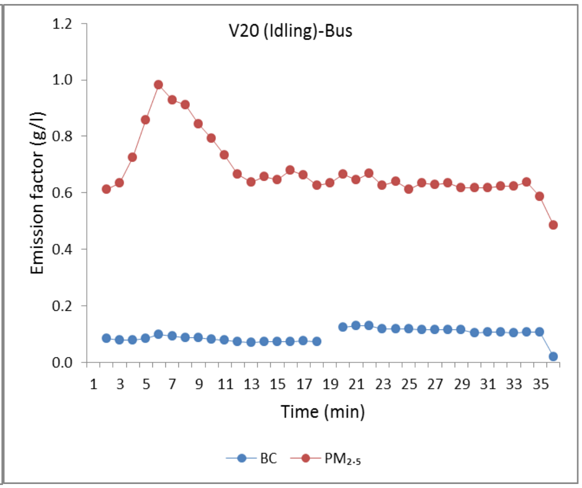

V18 (Idling)-Tourist Microbus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12
Time (min)
CO₂
CO

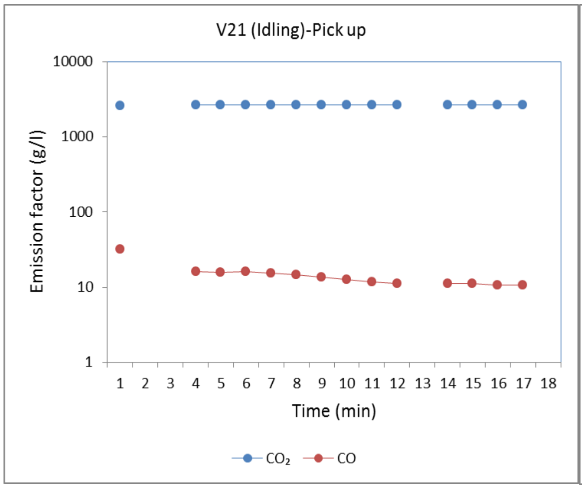

V18 (Idling)-Tourist Microbus
Emission factor (g/l)
8
7
6
5
4
3
2
1
0
1 2 3 4 5 6 7 8 9 10 11 12
Time (min)
BC
PM₂.₅

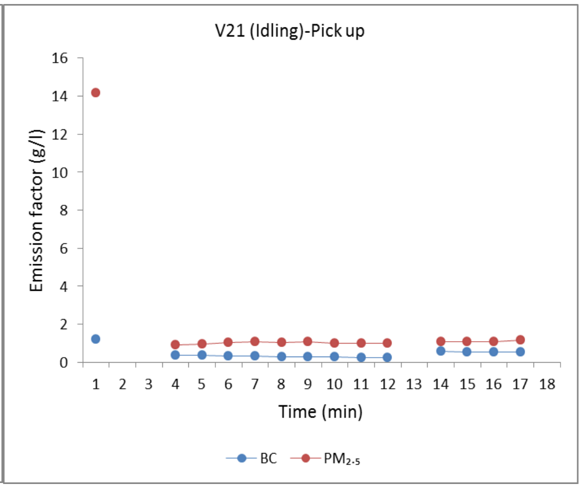

V19 (Idling)-Tourist bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
CO₂
CO

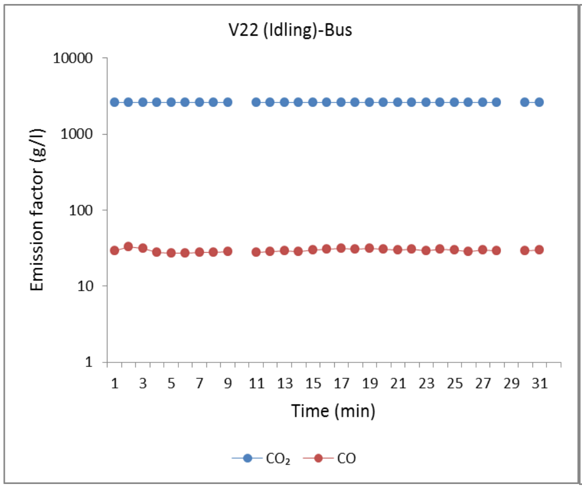

V19 (Idling)-Tourist bus
Emission factor (g/l)
12
10
8
6
4
2
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
BC
PM2.5

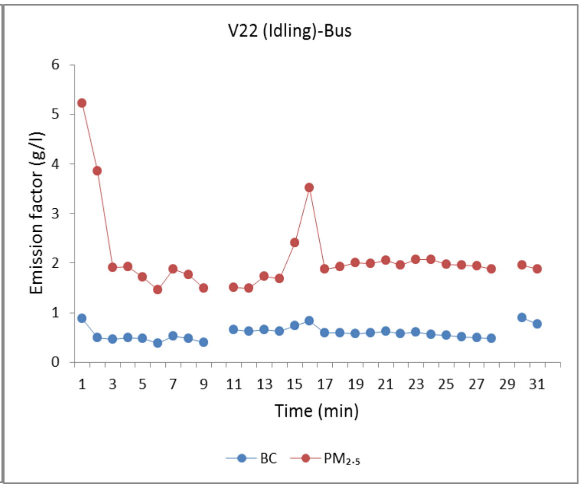

V20 (Idling)-Bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33 35
Time (min)
CO₂
CO

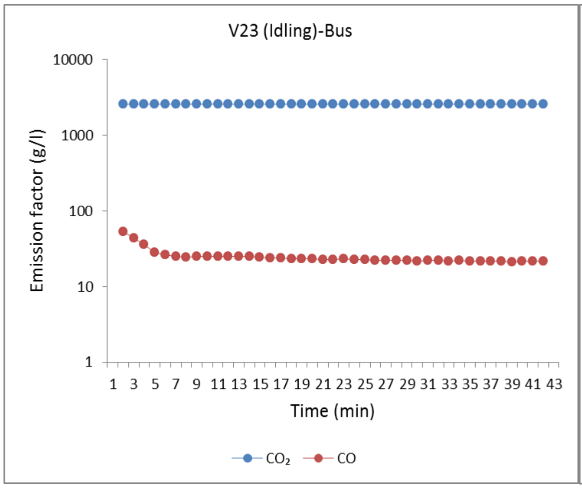

V20 (Idling)-Bus
Emission factor (g/l)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33 35
Time (min)
BC
PM2.5

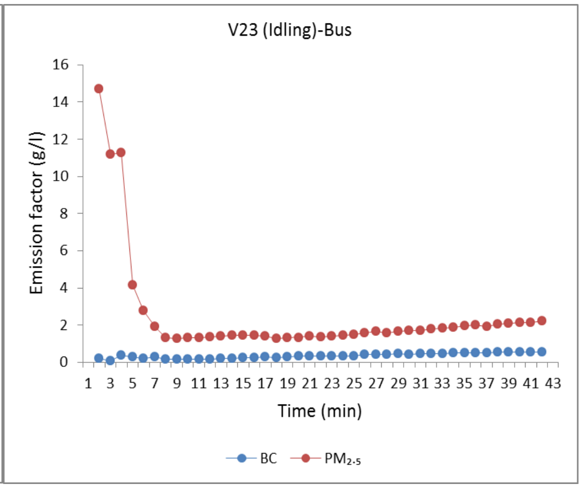

V21 (Idling)-Pick up
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18
Time (min)
CO₂
CO

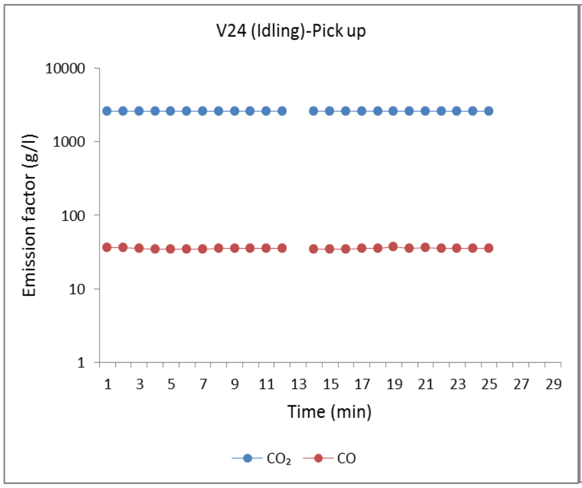

V21 (Idling)-Pick up
Emission factor (g/l)
16
14
12
10
8
6
4
2
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18
Time (min)
BC
PM2.5

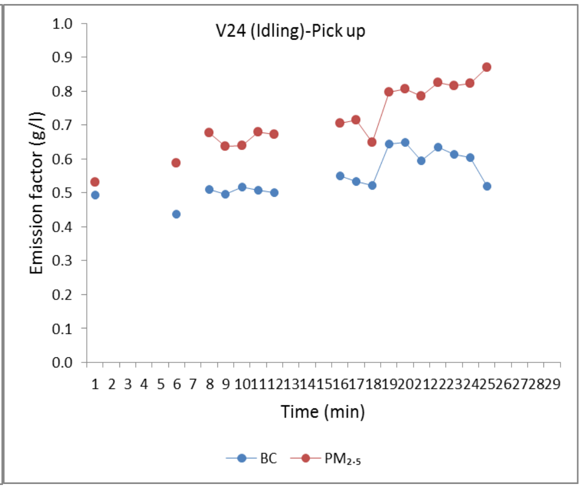

V22 (Idling)-Bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
CO₂
CO

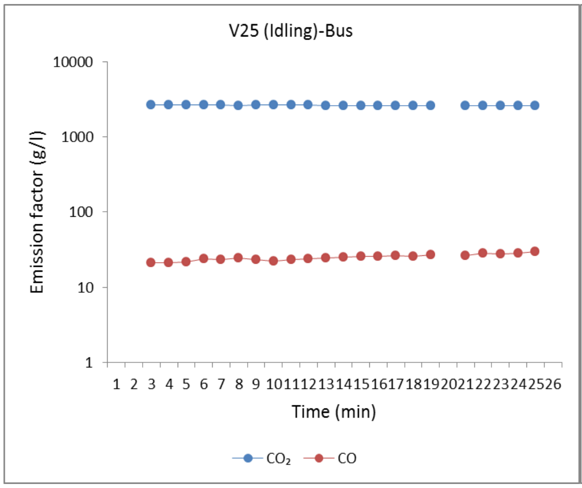

V22 (Idling)-Bus
Emission factor (g/l)
6
5
4
3
2
1
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
BC
PM2.5

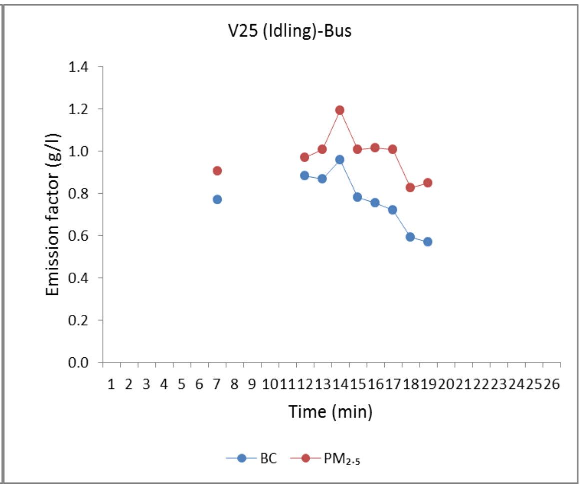

V23 (Idling)-Bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33 35 37 39 41 43
Time (min)
CO₂
CO

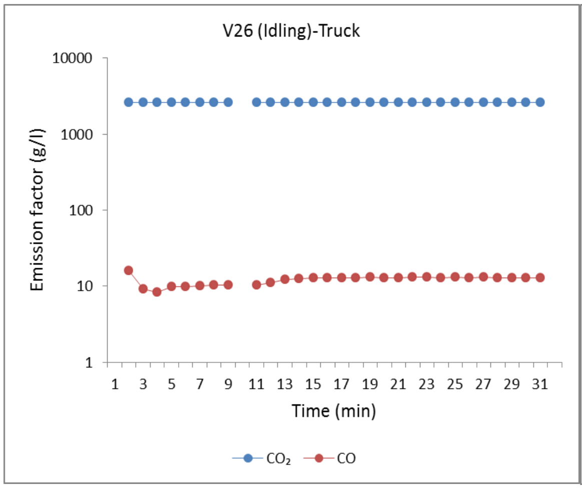

V23 (Idling)-Bus
Emission factor (g/l)
16
14
12
10
8
6
4
2
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33 35 37 39 41 43
Time (min)
BC
PM2.5

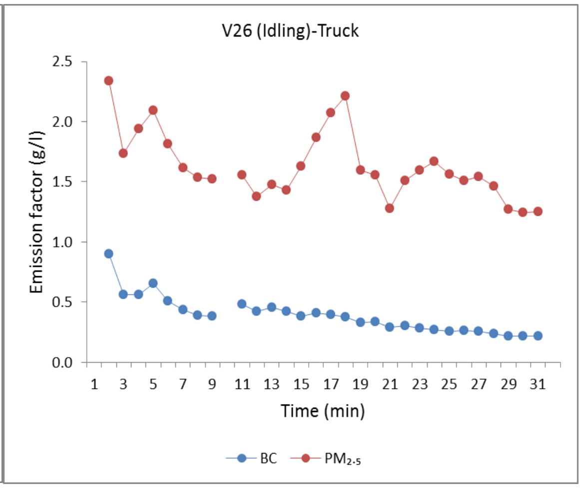

V24 (Idling)-Pick up
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29
Time (min)
CO₂
CO

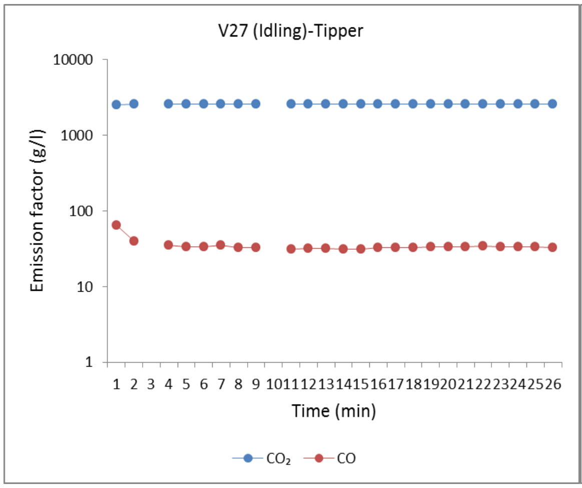

V24 (Idling)-Pick up
Emission factor (g/l)
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29
Time (min)
BC
PM2.5

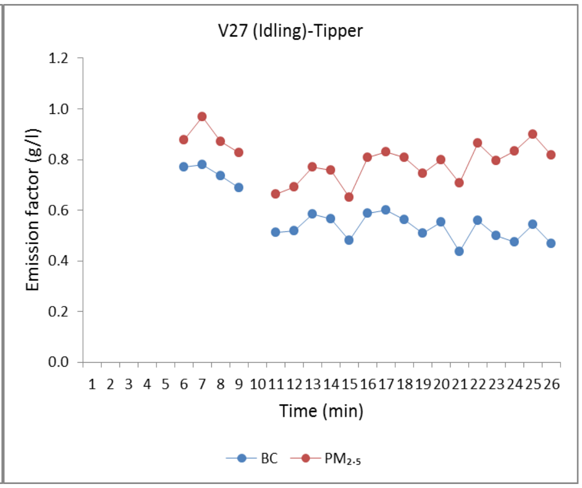

V25 (Idling)-Bus
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26
Time (min)
CO₂
CO

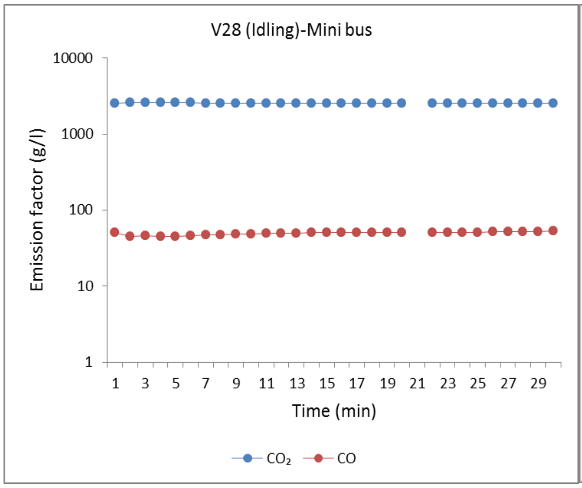

V25 (Idling)-Bus
Emission factor (g/l)
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26
Time (min)
BC
PM₂.₅

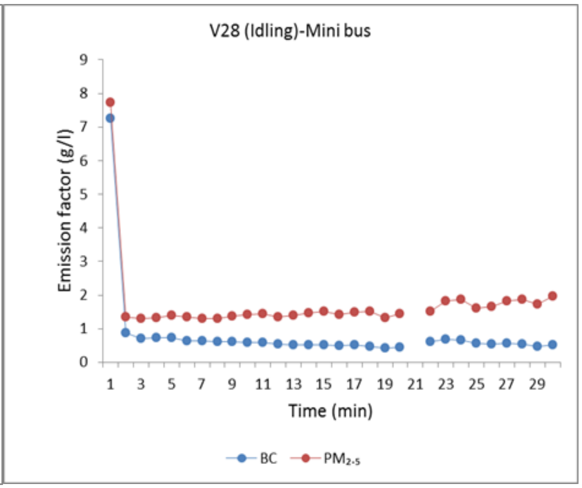

V26 (Idling)-Truck
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
CO₂
CO

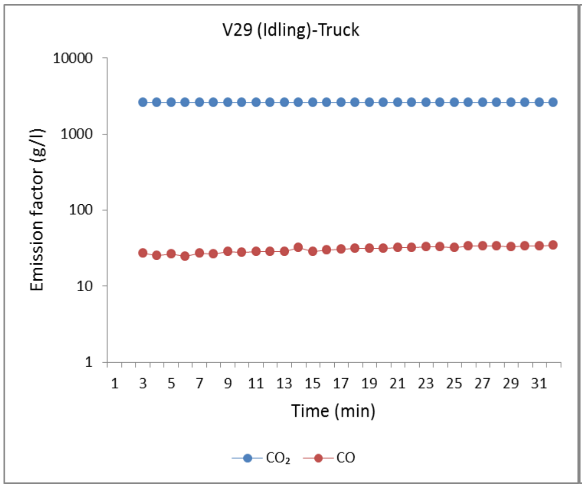

V26 (Idling)-Truck
Emission factor (g/l)
2.5
2.0
1.5
1.0
0.5
0.0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
BC
PM₂.₅

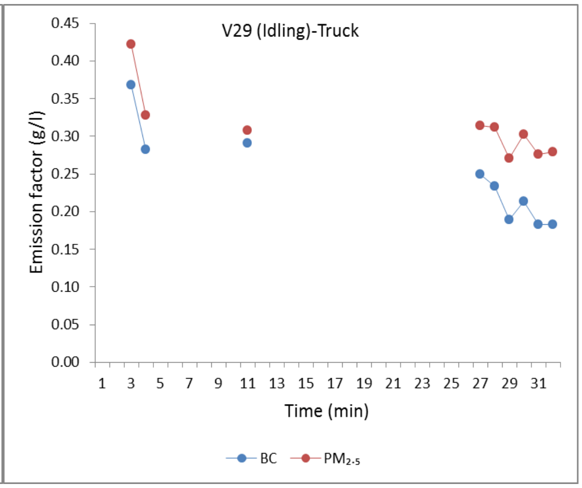

V27 (Idling)-Tipper
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26
Time (min)
CO₂
CO

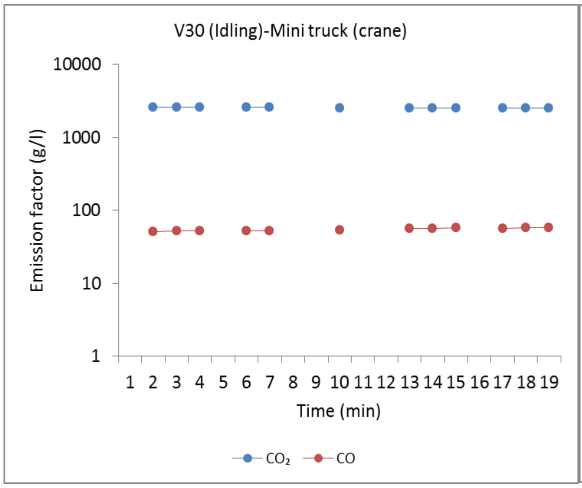

V27 (Idling)-Tipper
Emission factor (g/l)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26
Time (min)
BC
PM₂.₅

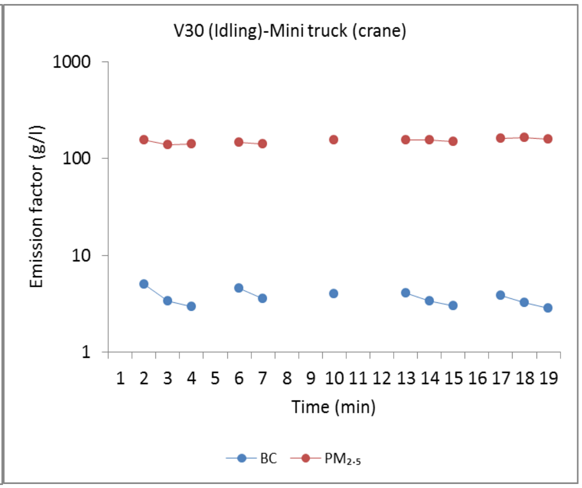

V28 (Idling)-Mini bus
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29
Time (min)
CO₂
CO

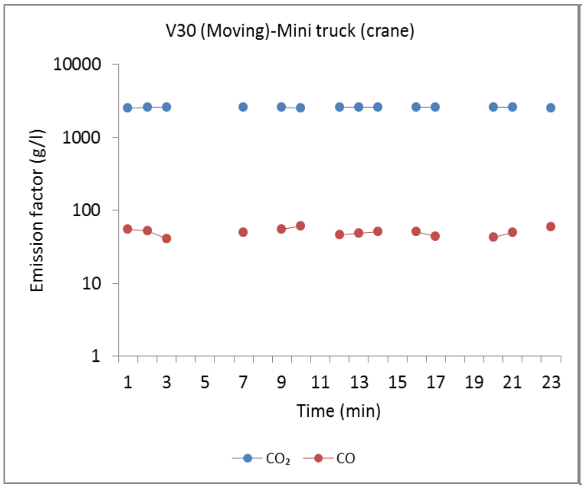

V28 (Idling)-Mini bus
Emission factor (g/l)
9
8
7
6
5
4
3
2
1
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29
Time (min)
BC
PM₂.₅

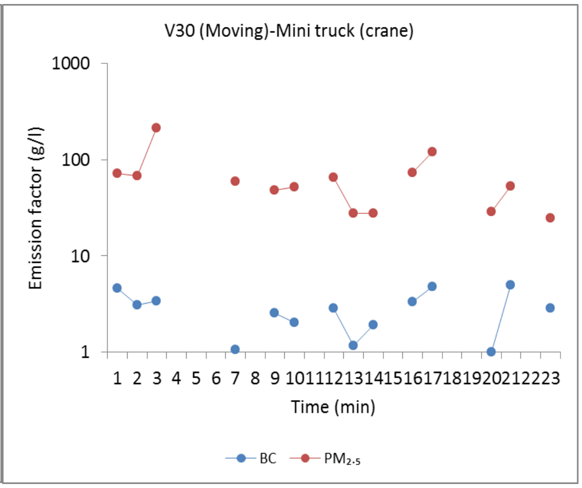

V29 (Idling)-Truck
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
CO₂
CO

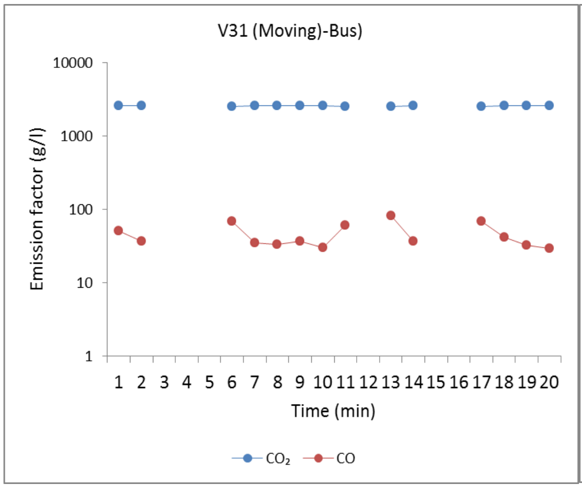

V29 (Idling)-Truck
Emission factor (g/l)
0.45
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31
Time (min)
BC
PM₂.₅

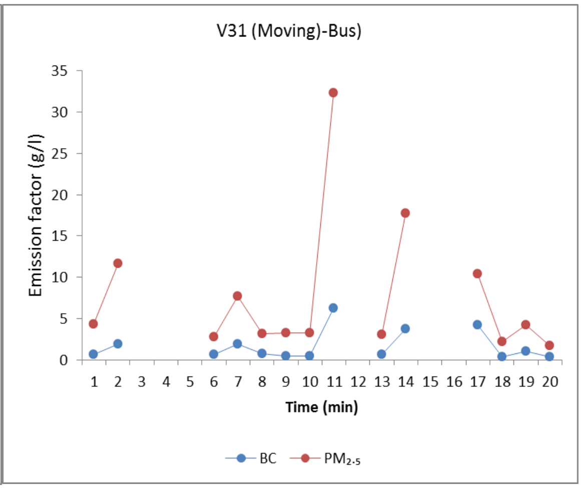

V30 (Idling)-Mini truck (crane)
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
CO₂
CO

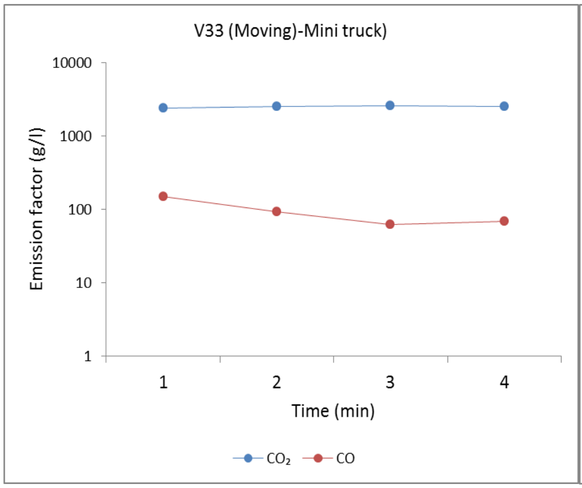

V30 (Idling)-Mini truck (crane)
Emission factor (g/l)
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Time (min)
BC
PM₂.₅

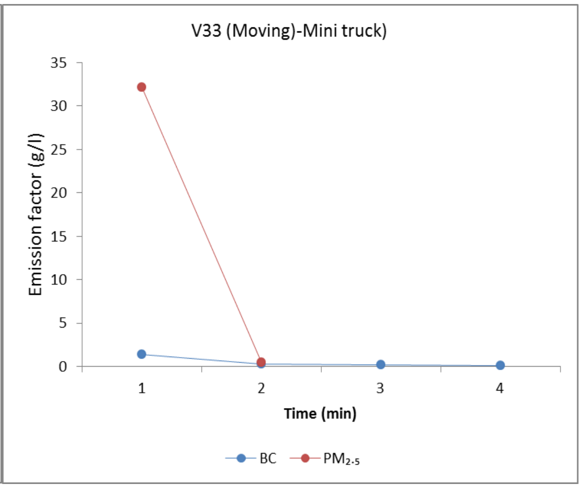

V30 (Moving)-Mini truck (crane)
Emission factor (g/l)
10000
1000
100
10
1
1 3 5 7 9 11 13 15 17 19 21 23
Time (min)
CO₂
CO

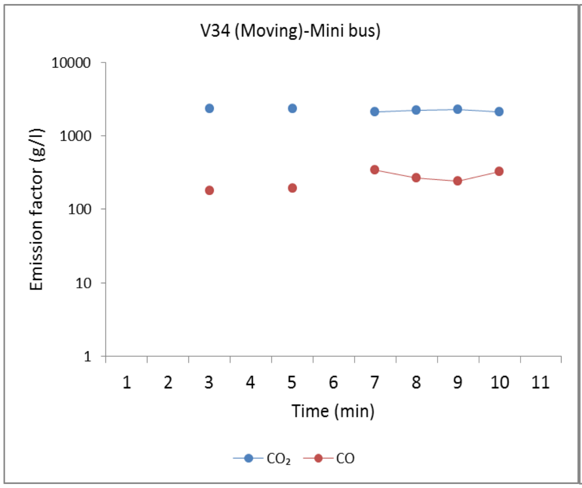

V30 (Moving)-Mini truck (crane)
Emission factor (g/l)
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23
Time (min)
BC
PM₂.₅

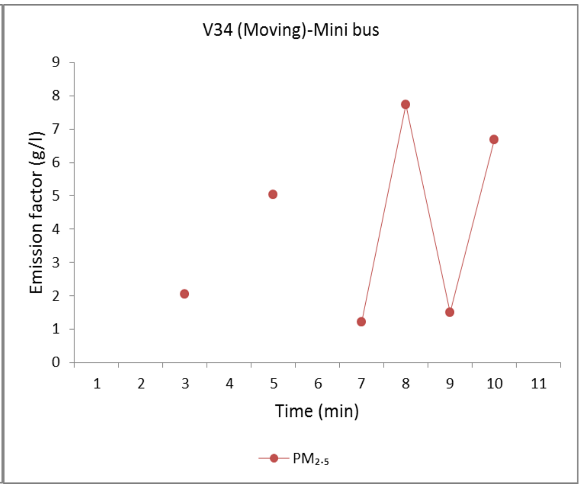

V31 (Moving)-Bus)
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20
Time (min)
CO₂
CO

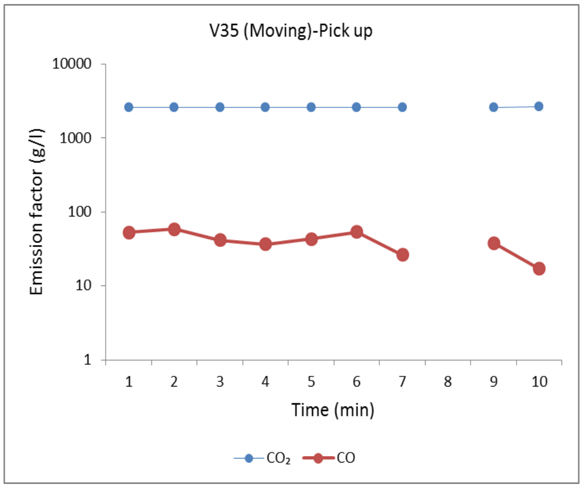

V31 (Moving)-Bus)
Emission factor (g/l)
35
30
25
20
15
10
5
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20
Time (min)
BC
PM₂.₅


V33 (Moving)-Mini truck)
Emission factor (g/l)
10000
1000
100
10
1
1 2 3 4
Time (min)
CO₂
CO


V33 (Moving)-Mini truck)
Emission factor (g/l)
35
30
25
20
15
10
5
0
1 2 3 4
Time (min)
BC
PM₂.₅


V34 (Moving)-Mini bus)
Emission factor (g/l)
10000
1000
100
10
1
1
2
3
4
5
6
7
8
9
10
11
Time (min)
CO₂
CO


V34 (Moving)-Mini bus
Emission factor (g/l)
9
8
7
6
5
4
3
2
1
0
1
2
3
4
5
6
7
8
9
10
11
Time (min)
PM2.5


V35 (Moving)-Pick up
Emission factor (g/l)
10000
1000
100
10
1
1
2
3
4
5
6
7
8
9
10
Time (min)
CO₂
CO

Appendix XX: Emissions Trend, Bagmati zone/Kathmandu Valley

	Idling condition (Gg)				**Moving condition (Gg)**			
Year	**CO_2 Max**	**CO Min**	**BC Min**	**$PM_{2.5}$ Min**	**CO_2 Min**	**CO Max**	**BC Max**	**$PM_{2.5}$ Max**
1989/90	39	1.5	0.0	0.1	38	5.1	0.2	1.0
1990/91	45	1.7	0.0	0.1	44	5.9	0.2	1.1
1991/92	51	1.9	0.1	0.2	49	6.6	0.2	1.3
1992/93	57	2.1	0.1	0.2	55	7.4	0.2	1.4
1993/94	72	2.7	0.1	0.2	70	9.3	0.3	1.8
1994/95	79	3.0	0.1	0.2	77	10.2	0.3	1.9
1995/96	87	3.2	0.1	0.3	84	11.2	0.4	2.1
1996/97	101	3.8	0.1	0.3	98	13.1	0.4	2.4
1997/98	110	4.1	0.1	0.3	107	14.2	0.5	2.6
1998/99	119	4.5	0.1	0.4	115	15.3	0.5	2.8
1999/00	129	4.6	0.1	0.4	124	15.8	0.5	3.0
2000/01	142	4.8	0.1	0.4	136	16.6	0.6	3.1
2001/02	158	5.1	0.1	0.4	151	17.4	0.6	3.3
2002/03	178	5.3	0.1	0.4	169	18.5	0.6	3.6
2003/04	194	2.7	0.0	0.3	184	9.1	0.5	3.3
2004/05	205	2.9	0.0	0.3	194	10.1	0.5	3.5
2005/06	225	3.1	0.1	0.3	212	10.8	0.5	3.8
2006/07	238	3.3	0.1	0.3	224	11.5	0.6	4.0
2007/08	260	3.6	0.1	0.4	245	12.5	0.6	4.4
2008/09	297	4.1	0.1	0.5	282	14.3	0.7	5.1
2009/10	356	5.0	0.1	0.6	337	16.7	0.8	6.0
2010/11	377	5.2	0.1	0.7	356	17.7	0.9	6.4
2011/12	392	5.4	0.1	0.8	370	18.5	0.9	6.6
2012/13	407	5.8	0.1	0.8	373	20.5	0.8	5.2
2013/14	439	6.1	0.1	0.8	405	21.0	0.8	5.4
2014/15	485	6.7	0.1	0.9	449	22.0	0.8	5.7
2015/16	533	7.5	0.1	0.9	496	22.9	0.8	5.8
2016/17	606	8.7	0.2	1.0	568	24.5	0.8	6.2
2017/18	679	9.7	0.2	1.0	637	26.2	0.8	6.5

Appendix XXI: Emissions Trend, Nepal

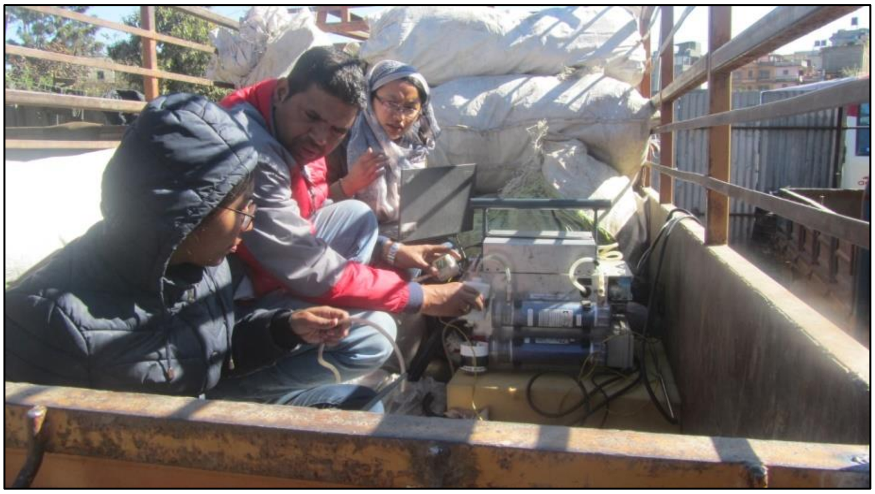
	Idling condition				**Moving condition**			
Year	**CO_2 Max**	**CO Min**	**BC Min**	**$PM_{2.5}$ Min**	**CO_2 Min**	**CO Max**	**BC Max**	**$PM_{2.5}$ Max**
1989/90	173	5.4	0.2	0.6	169	16.8	0.7	3.1
1990/91	196	6.2	0.2	0.6	192	19.2	0.8	3.6
1991/92	231	7.4	0.2	0.8	226	22.9	1.0	4.3
1992/93	264	8.4	0.3	0.9	258	26.0	1.1	4.8
1993/94	306	9.6	0.3	1.0	300	29.5	1.3	5.5
1994/95	343	10.7	0.3	1.1	336	32.7	1.4	6.1
1995/96	367	11.4	0.4	1.2	360	35.0	1.5	6.5
1996/97	391	12.1	0.4	1.3	383	37.1	1.6	6.9
1997/98	424	13.1	0.4	1.4	415	39.8	1.8	7.4
1998/99	450	13.8	0.4	1.5	441	41.8	1.9	7.8
1999/00	471	14.0	0.4	1.5	460	42.7	1.9	8.1
2000/01	511	14.5	0.5	1.6	494	44.3	2.0	8.6
2001/02	560	15.1	0.5	1.6	535	46.2	2.2	9.2
2002/03	596	15.5	0.5	1.7	566	48.1	2.2	9.6
2003/04	607	7.9	0.2	0.8	556	27.9	1.7	7.9
2004/05	623	8.1	0.2	0.8	569	29.5	1.7	8.0
2005/06	656	8.5	0.2	0.9	600	31.0	1.8	8.5
2006/07	712	9.2	0.2	0.9	651	33.7	1.9	9.2
2007/08	776	10.1	0.2	1.1	712	36.9	2.1	10.1
2008/09	840	10.9	0.2	1.2	770	39.9	2.3	10.9
2009/10	1076	14.0	0.3	1.5	990	51.0	2.9	14.2
2010/11	1150	15.0	0.3	1.8	1057	54.2	3.1	15.2
2011/12	1197	15.6	0.3	2.0	1101	56.6	3.2	15.9
2012/13	1293	16.5	0.3	2.1	1198	58.8	3.2	16.5
2013/14	1419	17.4	0.3	2.2	1288	60.8	3.2	17.0
2014/15	1628	19.1	0.4	2.4	1441	65.1	3.2	18.1
2015/16	1958	21.7	0.4	2.7	1671	72.6	3.3	19.8
2016/17	2385	24.8	0.5	3.1	1954	80.8	3.4	21.7
2017/18	2781	27.7	0.5	3.4	2214	88.8	3.6	23.5

Appendix XXII: Photographs of vehicular emission measurement and laboratory works

Filter paper preparation for field work

Emission measurement of pickup

Emission measurement of truck

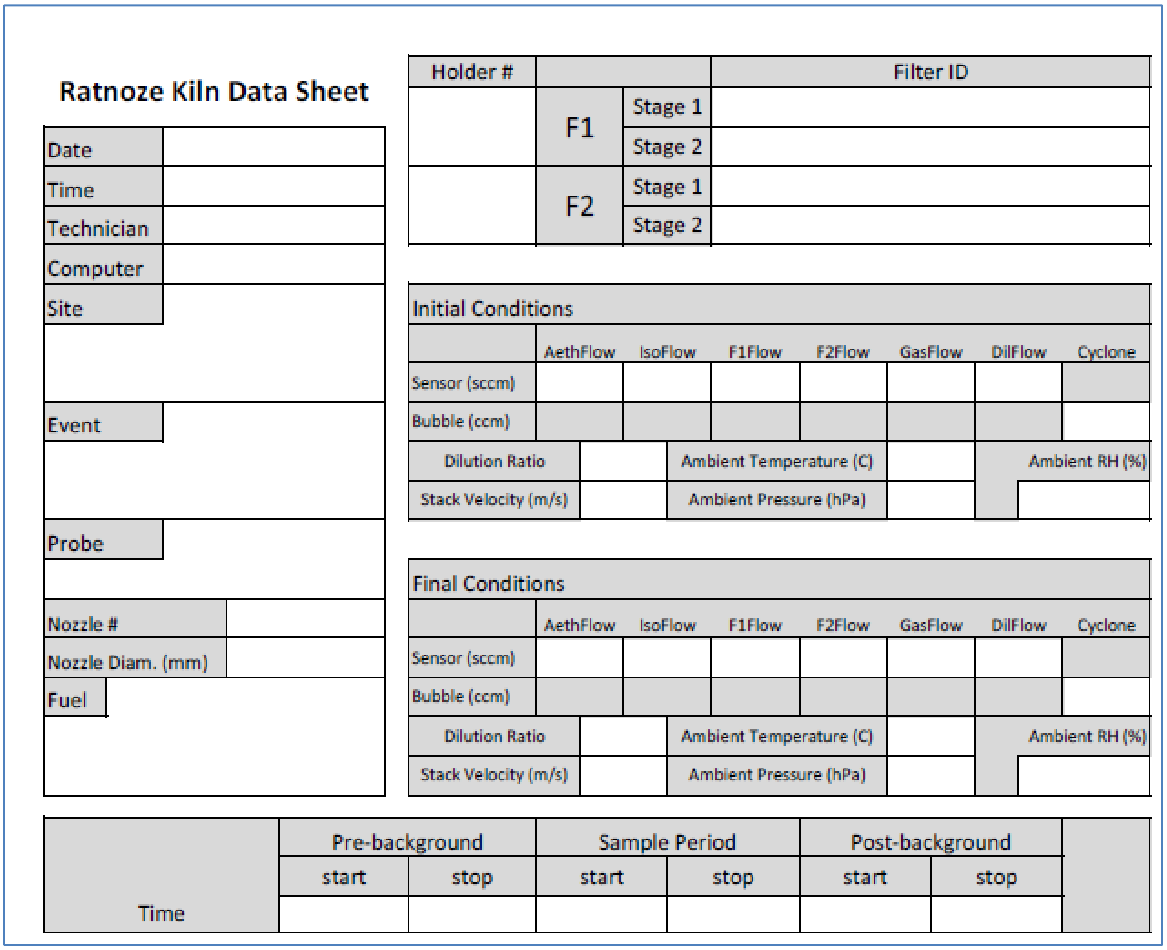


Emission measurement of truck

Emission measurement of bus

Filter paper mass measurment after field work

Appendix XXIII: Data sheet of Ratnoze that was used while vehicle emission measurement (Mountain Air Engineering)



Ratnoze Kiln Data Sheet

Date	
Time	
Technician	
Computer	
Site	
Event	
Probe	
Nozzle #	
Nozzle Diam. (mm)	
Fuel	

Holder #			Filter ID
	F1	Stage 1	
		Stage 2	
	F2	Stage 1	
		Stage 2	

Initial Conditions

	AethFlow	IsoFlow	F1Flow	F2Flow	GasFlow	DilFlow	Cyclone
Sensor (sccm)							
Bubble (ccm)							

Dilution Ratio		Ambient Temperature (C)		Ambient RH (%)
Stack Velocity (m/s)		Ambient Pressure (hPa)		

Final Conditions

	AethFlow	IsoFlow	F1Flow	F2Flow	GasFlow	DilFlow	Cyclone
Sensor (sccm)							
Bubble (ccm)							

Dilution Ratio		Ambient Temperature (C)		Ambient RH (%)
Stack Velocity (m/s)		Ambient Pressure (hPa)		

	Pre-background		Sample Period		Post-background		
	start	stop	start	stop	start	stop	
Time							

Appendix XXIV: Certificate of recognition from International Conferences/Workshops/Trainings

Newcastle University

This is to certify that

Bhupendra Das

has presented a paper in the

2nd Indian International Conference on

"Air Quality Management (IICAQM2017)"

jointly organized by the

Department of Civil Engineering, IIT Madras, Department of Civil Engineering, IIT Delhi and Newcastle University, UK on 1st and 2nd June 2017 at IIT Delhi

Dr. S. M. Shiva Nagendra
Chair IICAQM 2017
IIT Madras

Dr. Anil Namdeo
Chair IICAQM 2017
Newcastle University

Prof. Mukesh Khare
Chair IICAQM 2017
IIT Delhi

AMERICAN EMBASSY

ICIMOD

Certificate of Participation

Presented to

Bhupendra Das

for participating in the poster presentation session at the international workshop and science policy dialogue on

Air Pollution, Climate and Health in Southern Asia and the Hindu Kush Himalaya

27–28 November 2017, Kathmandu, Nepal

David Molden
Director General
ICIMOD

Arnico Panday
Regional Programme Manager (Interim)
Atmosphere Initiative

Certificate
of Attendance

No. : KP. 011/002/KB/VIII/2016

Organized by:

The Agency for Meteorology Climatology and Geophysics (BMKG) & World Meteorological Organization (WMO)

Supported by:

UNISDR IWPM International Wildfire Preparedness Mechanism

IBBI

UNITED NATIONS UNIVERSITY

BMKG

Jakarta, 29 August - 1 September 2016

This is to certify that

BHUPENDRA DAS

has successfully completed the

International Workshop on Biomass Burning

"Forecasting Emissions from Vegetation Fires and their Impacts on Human Health and Security in South East Asia"

as PARTICIPANT

Dr. Andi Eka Sakya, M.Eng
Director General of BMKG

Prof. Dr. Sci. Alexander Baklanov, PhD
Research Department, WMO

Certificate
of Attendance

No.: KP.011/06/D2/VIII/2016

Organized by:
The Agency for Meteorology Climatology and Geophysics (BMKG) & World Meteorological Organization (WMO)
Supported by:

Jakarta, 29 August - 1 September 2016

This is to certify that

BHUPENDRA DAS

has successfully completed the

Training on Biomass Burning
"Forecasting Emissions from Vegetation Fires and their Impacts on Human Health and Security in South East Asia"

as PARTICIPANT

Drs. R. Mulyono R. Prabowo, M.Sc
Deputy Director General for Climatolgy, BMKG

Prof. Dr. Sci. Alexander Baklanov, PhD
Research Department, WMO

Newcastle University

This is to certify that
Bhupendra Das
(Tribhuvan University, Nepal)
has participated in the
Summer School on
"Urban Air Pollution and Human Exposure Assessment"
jointly organized by the Department of Civil Engineering, IIT Madras, Department of Civil Engineering, IIT Delhi and Newcastle University, UK
from 29th to 31st May, 2017 at IIT Delhi.

Dr. S. M. Shiva Nagendra
Chair IICAQM 2017
IIT Madras

Dr. Anil Namdeo
Chair IICAQM 2017
Newcastle University

Prof. Mukesh Khare
Chair IICAQM 2017
IIT Delhi

Certificate of Participation

ICIMOD

Bhupendra Das

has successfully participated in the
Regional Workshop on
Air Quality Instrument Operation and Maintenance
21–25 November, 2016, Kathmandu, Nepal

David Molden
Director General
ICIMOD

Radheshyam Yadav
Sr Manager-Customer Care
Thermo Fisher Scientific India Pvt. Ltd.

Arnico Panday
Programme Coordinator
Atmosphere Initiative, ICIMOD

Appendix XXV: Photographs of International Conferences/Workshops/Trainings


Paper presentation at international conference, Delhi


Award for second best poster presentation, Delhi


Participation at international workshop, Delhi


Paper presentation at MoChWo conference, Nepal


International training, Jakarta, Indonesia


Instrument handling training, ICIMOD, Nepal

Appendix XXVI: Scientific papers published based on the research

Articles 1 (IF: 8.071)

Das, B., Bhave, P. V., Puppala, S. P., Shakya, K., Maharjan, B., and Rejina M. Byanju, R. M., 2020. A model-ready emission inventory for crop residue open burning in the context of Nepal. *Environmental Pollution*, **266**: 115069. https://doi.org/10.1016/j.envpol.2020.115069

Articles 2 (IF: 7.145)

Das, B., Bhave, P. V., Sapkota, A., Byanju, R. M. 2018. Estimating emissions from open burning of municipal solid waste in municipalities of Nepal. *Waste Management*, **79**: 481-490. https://doi.org/10.1016/j.wasman.2018.08.013

Articles 3 (IF: 7.963)

Das, B., Bhave, P.V., Puppala, S.P., Adhikari, S., Khanal, N., Mool, E., & Byanju, R.M., 2022. Emission factors and emission inventory of diesel vehicles in Nepal. *Science of the Total Environment*, **812**: 152539. https://doi.org/10.1016/j.scitotenv.2021.152539

Articles 4 (IF: 3.337)

Mool, E., Bhave, P.V., Khanal, N., Byanju, R.M., Adhikari, S., **Das, B.** and Puppala, S.P., 2019. Traffic condition and emission factor from diesel vehicles within the Kathmandu Valley. *Aerosol and Air Quality Research*, **20**(3): 395-409. https://doi.org/10.4209/aaqr.2019.03.0159

Articles 5 (Tribhuvan University, IoST)

Das, B., Bhave, P. V., Puppala, S. P., and Rejina M. Byanju, R. M., 2018. A global perspective of vehicular emission control policy and practices: an interface with Kathmandu Valley case, Nepal. *Journal of Institute of Science and Technology*, **23**(1):76-80. https://doi.org/10.3126/jist.v23i1.22199

Appendix XXVII: Other research related activities

International Proceedings

1. EPA (https://www.epa.gov/sites/production/files/2017-11/documents/resolution_technology.pdf)
2. Journal of Environmental Research (https://www.imedpub.com/proceedings/topdown-estimation-of-emissions-from-open-waste-burning-in-nepal-3448.html)

As an International Peer-Reviewer

1. SLOCAT Transport and Climate Change Global Status Report-2020, 2nd edition

Published date: June, 2021

[This study was supported by over 20 international development partners]

Available Link:

https://tcc-gsr.com/wp-content/uploads/2021/06/Slocat-Global-Status-Report-2nd-edition_high-res.pdf?fbclid=IwAR3ylEa1QB30mzaXhjvIqhreYmfaMPSwQ2rHfnHn-bv8x0k32-rroEK6wW4

2. Waste Management & Research: The Journal for a Sustainable Circular Economy, SAGE Publishing, USA (IF: 3.549)

International publication

1. Thakuri, S., Rijal, K., Pantha, R., Joshi, S., and **Das, B., 2017**. Mitigation potentials of GHG emissions from biomass burning in Nepal. *Climate Change and Green Growth Journal of GIR Korea*.
2. Neupane, D., **Das et al.** (2021). Growing Jatropha (Jatropha curcas L.) as a Potential Second-Generation Biodiesel Feedstock. J. Inventions. Available at https://www.mdpi.com/2411-5134/6/4/60

International award received:

10 & 11-Sep-2021: Received **"Young Scientist Award"** in the International Scientist Awards on Engineering, Science, and Medicine, held at Coimbatore, India. Awarded by VDGOOD Professional Association.

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A model-ready emission inventory for crop residue open burning in the context of Nepal☆

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ABSTRACT

Open burning of crop residue is an important source of air pollution which is poorly characterized in South Asia. Currently, the gridded inventory reported by Global Fire Emissions Database for biomass burning including open burning of crop residue are of coarse resolution (0.25° × 0.25°), and may not be appropriate for a simulation for Nepal. This study develops a comprehensive high resolution (1 km × 1 km) gridded model-ready emissions inventory for Nepal to understand the spatial characteristics of air pollutant emissions from open burning. We estimate the national air pollutant emissions from crop residue burned between the years 2003 and 2017. The best available data on agricultural production, residue consumption patterns, agricultural burning parameters and emission factors were derived from secondary sources. The Monte Carlo method was used to estimate uncertainties. The mass of crop residue burned in 2016/17 was 2908 Gg (61–139%), which was 22% of the dry matter generated that year. By multiplying the burned crop residue mass by emission factors, the air pollutant emissions were estimated as 4140 for CO_2 (56–144%), 154 for CO (4–196%), 6.5 for CH_4 (7–193%), 1.2 for SO_2 (60 −140%), 24.5 for $PM_{2.5}$ (30–170%), 8.6 for OC (38–162%), 2.2 for BC (-1-201%), 7 for NO_x (54–146%), 22.5 for NMVOC (8–192%) and 2.7 for NH_3 (3–197%) in unit of Gg yr^{-1}. More than 80% of air pollutants were generated during the months of February to May from the open burning of crop residue. The findings of this paper indicate that substantial reduction in open field burning would dramatically improve air quality in both the Terai region and other parts of Nepal and help reduce negative health impacts associated with the open burning of residue such as premature deaths, respiratory disease, and cardiovascular disease.

1. Introduction

Crop residue open burning (CROB) has become a type of biomass burning (Streets et al., 2003) causing much concern around the world. It is practiced in many countries especially during the harvesting season (Chang et al., 2013). Crop residue is the fourth largest fuel after coal, oil and natural gas. Half of the global population (mostly in developing countries) use it for space heating and cooking (Guoliang et al., 2008). A plethora of studies on CROB have attempted to understand its effects on the atmosphere and climate (Venkataraman et al., 2006). In Asia, intensive paddy straw open burning is prevalent during the dry season, contributing in large measure to ambient air pollution (Tipayarom and Oanh, 2007). Burning of crop residue not only releases known air pollutants (NO_X, SO_2, CO, NMVOC, NH_3) but also greenhouse gases (CO_2, CH_4, N_2O) and particulate matter (PM_{10}, $PM_{2.5}$, BC, OC) (Gadde et al., 2009; Shrestha, 2018; Chang and Song, 2010; Zhang et al., 2000; Andreae and Merlet, 2001; Tipayarom and Oanh, 2007; Hossain and Park, 2012; Viana et al., 2008; Dennis et al., 2002; Sahai et al., 2007). After the air pollutants are released into the atmosphere, they undergo further chemical and physical transformations which have the potential to alter the earth's radiation balance and adversely affect air quality (Jethva et al., 2019; Permadi

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and Oanh, 2013) and human health (Tipayarom and Oanh, 2007; Zhang et al., 2017). Unlike many industrial point sources, the impact of residue burning can be widespread (Jethva et al., 2019). A recent study conducted in India showed that residue burning coupled with the annual festival of lights (Diwali) impacted Delhi's weather in the form of fog, haze and smog that lasted for a couple of weeks (i.e., October to November 2016) leading to reduced atmospheric visibility (Chauhan and Singh, 2017) and temporary school closures. The air quality in Delhi between October and November 2019 has been the worst on record so far. There are serious health outcomes associated with residue burning. According to a study conducted in Japan, emissions due to paddy straw burning have led to increased asthma attacks among children (Torigoe et al., 2000). Similarly, a study in China has suggested that $PM_{2.5}$ exposure resulted in USD 8822.4 million health related economic loss just from straw open burning (Li et al., 2017). This is not all. Air pollution indirectly affects the economy of a country as well.

Although several studies have been conducted on CROB emissions worldwide (Li et al., 2017; Ni et al., 2015; Gadde et al., 2009; Streets et al., 2003; Irfan et al., 2014; Sahai et al., 2011; Andreae and Merlet, 2001; Yang et al., 2008; Zhang et al., 2008; Shrestha, 2018), gridded model-ready emission inventories are limited (Pandey et al., 2014; Venkataraman et al., 2006; Chang and Song, 2010; Chang et al., 2013). In addition, high uncertainties and large information gaps exist in the estimation of emissions from open burning of residue (Chang et al., 2013; Gadde et al., 2009; Mona and Hitesh, 2016). The current approach adopted in studies to estimate the fraction of crop residue burnt in the agricultural fields is also not well established, resulting in further uncertainties (Venkataraman et al., 2006).

Nepal is a small, landlocked country which occupies an area of 147,181 sq. km of which only 28% constitute agricultural land; however, agriculture sector provides a livelihood to more than 64% of the population and contributes about one third of the national GDP (Central Bureau of Statistics, 2014; Ministry of Agriculture Development, 2015). While part of the crop waste generated is used as animal fodder and fuel and for thatching roofs, the remaining residue is burnt in the field (Sahai et al., 2011).

A model-ready emission inventory requires the utilization of more sophisticated tools such as Geographical Information System (GIS) (Markakis et al., 2013). To determine global biomass burning, remote sensing is also applied though it has not been used widely in Asian countries yet (Chang et al., 2013). There are important caveats and limitations that should be considered when using a satellite-based data set as the size of the agricultural land and active fire spots that are considerably smaller than the spatial scale of satellites with 1 km^2 resolutions lead to mixed pixel problems in image classification (Chang et al., 2013). Moreover, satellite repeat cycles are unable to detect short-lived crop fires and in such instances active fire data may not be accounted, which would lead to high uncertainty in the result (Kanabkaew and Oanh, 2011; VanderWerf et al., 2010). A study by Benali et al. (2016) has suggested that active fires of short duration (<12 h) are not detected by orbital satellites. Furthermore, it revealed persistent cloud cover as well as multiple fire events to be in a single fire perimeter which causes further uncertainties. The Global Fire Emissions Database (GFED), though a widely used emissions inventory, is unable to cover the data sets of many smaller fires (Zhang et al., 2018), which contributes to the uncertainties.

This study attempts to present the air pollutants profile resulting from CROB for 2016/17 and projected between 2003/04 and 2016/17 for Nepal. Although, at present, very coarsely resolved (0.25° by 0.25°) gridded inventories (Kurokawa et al., 2013; Rupakheti et al., 2017; Bhardwaj et al., 2018) and at high resolution (1 km × 1 km) (Sadavarte et al., 2019) are available, they exclude the agricultural burning sector. This study attempts to develop a fine-resolution, 1 km × 1 km gridded version for the year 2016/17. It would be the first study nationally to develop a very robust emission inventory for open burning of residue.

2. Data and methods

2.1. Crop production

The district-wise crop production data from 2003/04 to 2016/17 (tons/year) were obtained from the Ministry of Agriculture Development (MoAD) (2003/04–2016/17) Annual Reports. The crops were categorized as paddy, maize, millet, wheat, barley, oil crops, potato, sugarcane, jute and pulses. Pulses included lentil, chick pea, pigeon pea, black gram, grass pea, horse pea, soybean and other types of crops. By adding up crop production in all 75 districts, the national crop production was estimated.

2.2. Residue generation

Residue to crop ratio was the factor which was used to calculate residue generation from crop production. Its values for the different crops was collected from those reported in the literature: for rice, Singh and Gu (2010), Sahai et al. (2011), and Jain et al. (2014); for maize, Singh and Gu (2010), Koopmans and Koppejan (1997), and Yang et al. (2008); for millet and wheat, Singh and Gu (2010) and Sahai et al. (2011); for barley, Singh and Gu (2010); for oil crops, Yang et al. (2008); for potato, Yang et al. (2008); for sugarcane, Singh and Gu (2010) and Global Atmospheric Pollution Forum (2010); for jute, Koopmans and Koppejan (1997); and for pulses, Yang et al. (2008) (see Table 1). The dry matter to crop residue ratio was the factor that was used to calculate the dry matter from residue generation. Its value for rice was taken from Streets et al. (2003); for maize, wheat, barley and potato from Intergovernmental Panel on Climate Change (IPCC). (1996); for millet from Global Atmospheric Pollution Forum. (2010); for oil crops from Jain et al. (2014) and Bhattacharya and Mitra (1998); for pulse from Shrestha (2018); and for sugarcane and jute from Bhattacharya and Mitra (1998) (see Table 1).

After multiplying the crop production amount by residue to crop ratio and dry matter to crop residue ratio, the total dry matter generated was calculated (Eq. (1)) (Kanabkaew and Oanh, 2011; Shrestha et al., 2013; Shrestha, 2018; IPCC, 1996).

$$C_R = P_1 \times N_1 \times D_1 \tag{1}$$

where.

C_R is the total dry matter/dry residue; P_1 is the crop production amount for crop type l; N_1 is the residue to crop ratio for crop type l; and D_1 is the dry matter to crop residue ratio for crop type l.

2.3. Amount of crop residue burnt in the field

The amount of crop residue available after its utilization pattern (i.e., fuel energy, roof thatching and animal fodder) for open burning (Eq. (2)) was taken from the available literature (Sahai et al., 2011). In districts where a residue consumption deficit prevailed, crop residue for open burning was estimated to be zero,

$$B_l = C_R - (C_F + C_T + C_A) \tag{2}$$

where B_l is the amount of crop residue available for open burning; C_F is the crop residue used as fuel energy; C_T is the crop residue used in thatching the roof; and C_A is the crop residue used as animal fodder. The available national data on fuel energy (in Gigajoule)

Table 1
Selected best value of parameters for emission estimation from the crop residue burning.

	Paddy	Maize	Millet	Wheat	Barley	Oil crops	Potatoes	Sugarcane	Jute	Pulses	Average
Residue to crop ratio (N)	1.5[a,b,c]	2[a,d,e]	1.2[a,b]	1.5[a,b]	1.3[a]	2[e]	0.5[e]	0.33[a,h]	2[i]	1.5[e]	–
Dry matter to crop residue ratio (D)	0.85[j]	0.4[k]	0.8[h]	0.83[k]	0.83[k]	0.8[c,l]	0.45[k]	0.8[l]	0.8[l]	0.8[g]	–
Burn efficiency fraction (η)	0.89[m]	0.92[m]	0.9[k]	0.86[m]	0.82[m]	0.9[k]	0.9[k]	0.68[m]	0.9[k]	0.9[g]	0.86 ± 0.07[n]

[a] Singh and Gu (2010).
[b] Sahai et al., 2011.
[c] Jain et al., 2014.
[d] Koopmans and Koppejan, 1997.
[e] Yang et al., 2008.
[g] Shrestha, 2018.
[h] GAPF, 2010.
[i] Koopmans and Koppejan, 1997.
[j] Streets et al., 2003.
[k] IPCC, 1996.
[l] Bhattacharya and Mitra (1998).
[m] Turn et al., 1997.
[n] calculation.

were compiled for the years 2003/04 to 2008/09 (Water and Energy Commission Secretariat, 2010) and 2014/15 (Water and Energy Commission Secretariat, 2017) and converted to metric tons by referring to Ministry of Agriculture Development (2014). For the remaining years (i.e., 2009/10–2013/14; 2015/16–2016/17), the fuel energy value was estimated based on linear regression. These national values were further distributed at the district level in line with the weightage of residue generation. The proportion of crop residue used for roof thatching in Nepal is not reported in the literature yet. Therefore, its usage was extrapolated from that in neighboring India on the basis that the values would be similar. Venkataraman et al. (2006) has reported that 2% of the total paddy residue generation is used for thatching houses in India. Hence, this was proportionately used to calculate the amount of residue used for roof thatching from 2003/04 to 2016/17 at the district level in Nepal. The animal census and feed requirement values were obtained from the available literature to estimate the residue used as animal fodder. The major domestic animal types like total cattle, total buffalo, milking cow and milking buffalo were considered for this study. The district-wise animal census for the years 2003/04 to 2016/17 was taken from the MoAD Annual Reports (2003/04–2016/17). The data on non-milking cattle and buffalo (i.e., dry, adult males and young stock) that are not reported was assumed to be in the same proportion as in India (Dikshit and Birthal, 2010). The complete cattle and buffalo census for Nepal was estimated on this. The daily dry matter requirement for cattle and buffalo (i.e., milking, dry, adult males and young stock) was obtained from Dikshit and Birthal (2010) (see Supplementary Table 1). The total dry matter requirement for the years 2003/04 to 2016/17 was calculated this way. The seasonal dry feed intake was obtained from National Animal Feed and Livestock Quality Management Laboratory (2019), the national animal feed inventory of Nepal.

After multiplying B_l by η (burn efficiency fraction), the mass of residue burnt (M_1) in the agricultural field was calculated (Kanabkaew and Oanh, 2011; Shrestha et al., 2013; Shrestha, 2018; IPCC, 1996).

$$M_l = B_l \times \eta \tag{3}$$

η is also defined as fraction of the dry matter exposed to burning that actually burns. It does not vary significantly from crop to crop. The values of η for rice, maize, wheat, barley, and sugarcane were obtained from Turn et al. (1997). Similarly, η for millet, oil crops, potato, tobacco and jute were compiled from IPCC (1996). η for pulses was taken from Shrestha (2018). The average value of η was then calculated, which was used in eq. (3), for estimating the mass of residue burnt openly (Table 1).

2.4. Emission factors and estimation of emissions

The study considered the best available country-specific emission factors (EFs) from the Nepal Ambient Monitoring and Source Testing Experiment (NAMaSTE) campaign (Stockwell et al., 2016; Jayarathne et al., 2018). In cases where country-based EFs were not available, values from other countries were taken (Streets et al., 2001; Andreae and Merlet, 2001; Dennis et al., 2002; Kanabkaew and Oanh, 2011; Turn et al., 1997; Cao et al., 2008). In most of the datasets where EFs reported were more than one value, they were averaged (see Table 2). Table 3 lists the literature from which they have been compiled. The average EFs and SD were calculated for each pollutant types from the selected crops (see Table 2 & Supplementary Fig. 1). These values were referred when calculating the emission trends. The most comprehensive district-wise emissions were presented for the year 2016/17.

Emission 'E' for pollutant type 'i' and crop type 'l' was estimated using eq. (4), where 'M_1' from eq. (3) was multiplied with emission factor '$EF_{i,l}$' specific to pollutant type 'i' from crop type 'l' (Kanabkaew and Oanh, 2011; Shrestha et al., 2013; Shrestha, 2018; IPCC, 2006).

$$E_{l,l} = \sum_{l} M_l \times EF_{l,l} \tag{4}$$

2.5. Spatial and temporal distribution

A model-ready emission inventory requires the utilization of more sophisticated tools such as Geographical Information System (GIS) and fire counts as proxy for temporal estimation (Markakis et al., 2013; Ni et al., 2015). This study used a computer-based model to develop a fine-resolution (1 km × 1 km) gridded emissions inventory for agricultural residue open burning for Nepal for the year 2016/17. It comprises the following datasets: Nepal's outline/boundary; Nepal's district boundary; land-cover of 2010 from International Centre for Integrated Mountain Development (2013) (see Supplementary Fig. 2); and district-wise agriculture production data for 2016/17. The emissions from CROB were based on the percentage of agriculture area under each grid by district. Nationally, spatial distributions were developed to understand the BC emission flux for better air quality management. For temporal distribution of emissions, the Visible Infrared Imaging Radiometer

Table 2
Selected best value of EFs for emission estimation from crop residue burning (g/kg).

	Rice	Maize	Millet	Wheat	Barley	Oil crops	Potatoes	Sugarcane	Jute	Pulse	Avg. EFs (SD)
CO_2	1474[q]	1322 ± 45[i]	1613[s]	1445[q]	1445[y]	1395[q]	1613[s]	1130[zb]	1345[ze]	1470[zl]	1425±(142)
CO	31.4[q]	60.4 ± 28.9[h]	36.4[u]	49.4[q]	49.4[y]	77.0[q]	36.4[c]	34.7[u]	106[ze]	47.3[zl]	52.8±(23.2)
CH_4	1.2[q]	3.0±2[j]	2.7[s]	1.9[q]	1.9[y]	3.0[q]	2.7[s]	0.4[u]	3.6 ± 1.3[zf]	1.9[zl]	2.2±(1.0)
SO_2	0.4[zj]	0.4[s]	0.4[s]	0.5[zk]	0.5[y]	0.4[s]	0.4s	0.4[s]	0.4[s]	0.4[y]	0.4±(0.02)
OC	4.2±3[c]	3.3 ± 1.8[m]	2.9 ± 0.5[v]	2.7 ± 1.5[f]	2.7 ± 1.5[y]	2.5 ± 0.8[z]	2.9 ± 0.5[v]	1.5[zc]	3.6 ± 2.2[zg]	3.3[s]	3.0±(0.7)
$PM_{2.5}$	11.1 ± 3.2[b]	8.5±4[n]	7.7 ± 3.8[w]	6.7 ± 3.0[e]	6.7 ± 3.0[y]	11.5[za]	7.7 ± 3.8[w]	3.8 ± 0.1[zd]	9.2 ± 4.7[zh]	11.5[zm]	8.4±(2.5)
BC	0.3[q]	0.7[o]	0.7[s]	0.6[q]	0.6[y]	1.7[q]	0.7[s]	0.6 ± 0.1[zc]	0.7[s]	0.8[zl]	0.8±(0.3)
NO_x	2.8 ± 0.9[a]	2.4 ± 1.6[p]	2.5 ± 0.6[x]	1.9 ± 0.9[d]	1.9 ± 0.9[y]	2.5 ± 0.6[x]	2.5 ± 0.6[x]	2.6[u]	2.6 ± 0.6[zi]	2.5[s]	2.4±(0.3)
NMVOC	7.0[s]	7.2±4[r]	11.1 ± 4.9[k]	4.8 ± 3.8[g]	4.8 ± 3.8[y]	11.1 ± 4.9[k]	11.1 ± 4.9[k]	2.2[u]	11.1 ± 4.9[k]	7.0[s]	7.7±(3.2)
NH_3	0.2[q]	0.9 ± 0.3[t]	1.3[s]	0.7[q]	0.7[y]	1.5[q]	1.3[s]	1.0[u]	1.3[s]	0.6[zl]	0.9±(0.4)

The values below are obtained from Table 3.
a = avg. from ab, ac and ad; b = avg. from ae, af, ag and ah; c = avg. from ae, af, ag, ah, zc and ze; d = avg. from u, ab, ab, ae, ai and ak; e = avg. from u, ae, ag, ah and ai; f = avg. from o, ae, ag, ah, ai, ak, zc and ze; g = avg. from u and ai; h = avg. from u, ab, ag, ai, an and ze; i = avg. from ab, ag, ai and an; j = avg. from u and ai; k = avg. from s and ai; m = avg. from ae, ag, ai, zc and ze; n = avg. from u, ae, ag and ai; p = avg. from ab, ai and u; r = avg. from u and ai; t = avg. from s, u and ai; v = avg. from s, ap and zc; w = avg. from s, ah, ap and zc; x = avg. from s, ak, ap, aq and ze; z = avg. from s, ae, ap and zc; za = avg. from s, ae, ah, ap, zc and zm; zd = avg. from u and zc; zf = avg. from s and zb; zg = avg. from s, ae, ap, zc and ze; zh = avg. from s, ae, ap, zb and zc; zi = avg. from s, ak, ap, aq and ze; zj = avg. from ac, an and ze; zk = avg. from ai, an and ze; zl = avg. value of q; zm = avg. value.

Table 3
Compiled sources of EFs for straw and residues.

[o] Streets et al. (2001)	BC (maize)	China (combined value from the literature)
[q] Stockwell et al. (2016)	CO_2, CO, CH_4, BC, NH_3 (rice, wheat, oil crops); average EFs of CO_2, CO, CH_4, BC, NH_3 (pulses)	Nepal (field experiment based value from NAMaSTE campaign)
[s] Andreae and Merlet, 2001	NMVOC (rice); SO_2 and NH_3 (maize); CO_2, CH_4, BC, NH_3, SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (millet and potato); SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (oil crops); SO_2 (sugarcane); SO_2, BC, NH_3, NMVOC (jute); OC, NO_x, NMVOC (pulses)	Global-based compiled value
[u] Dennis et al. (2002)	CO, CH_4, $PM_{2.5}$, NO_x, NMVOC, NH_3 (maize); CO (millet and potato); $PM_{2.5}$, NO_x, NMVOC (wheat); CO, CH_4, NO_x, NMVOC, NH_3, $PM_{2.5}$ (sugarcane); OC, $PM_{2.5}$ (jute)	United States
[y] best assumption	–	–
[ab] Zhang et al. (2008)	NO_x (rice); CO_2, CO, NO_x (maize); NO_x (wheat); $PM_{2.5}$, NO_x (jute)	China (laboratory based value)
[ac] Irfan et al. (2014)	SO_2, NO_x (rice)	Pakistan (field experiment based value)
[ad] Guoliang et al. (2008)	NO_x (rice)	Rural China (laboratory based value using a combustion chamber)
[ae] Li et al. (2017)	OC, $PM_{2.5}$ (rice, maize, wheat, oil crops); CH_4 (jute)	China (laboratory based value)
[af] Oanh et., 2011	OC, $PM_{2.5}$ (rice)	Thailand (in-situ experiment and laboratory hood experiment)
[ag] Ni et al. (2015)	OC, $PM_{2.5}$ (rice, wheat); CO_2, CO, OC, $PM_{2.5}$ (maize); CH_4, $PM_{2.5}$, NO_x (jute).	China (laboratory based value using a combustion chamber)
[ah] Hays et al. (2005)	OC, $PM_{2.5}$ (rice, wheat); $PM_{2.5}$ (millet, potato, oil crops); CH_4 (jute)	United States (experiment based values)
[ai] Li et al. (2007)	CO_2,CO, CH_4, SO_2, OC, $PM_{2.5}$, NO_X, NMVOC, NH_3 (maize); NMVOC (millet and potato); SO_2, OC, $PM_{2.5}$, NO_x, NMVOC (wheat); NMVOC (oil crops); CH_4, OC, $PM_{2.5}$, NO_x, NMVOC (jute)	Rural China (experimental value using middle volume samplers)
[ak] Sahai et al. (2007)	NO_x (millet, potato, oil crops); OC, NO_x (wheat); CH_4 (jute)	India (in-situ experiment of wheat straw burning in a combine harvested agricultural field)
[al] Zhang et al. (2000)	NO_x (wheat)	Rural China (experiment based on residue fuel and stove combination)
[an] Jenkins et al. (1996)	SO_2 (rice, wheat); CO_2, CO (maize); $PM_{2.5}$, NO_x (jute)	United States (wind tunnel simulation)
[ap] Akagi et al., 2011	OC, $PM_{2.5}$, NO_x (millet, potato, oil crops)	Global-based (laboratory based value through flaming and smoldering)
[aq] de Zarate et al. (2001)	NO_x (millet, potato, oil crops)	Spain (combustion chamber experiment)
[zb] Kanabkaew and Oanh (2011)	CO_2 (sugarcane)	Thailand
[zc] Turn et al. (1997)	OC (rice, maize, wheat); OC, $PM_{2.5}$ (millet, potato, oil crops); OC, BC, $PM_{2.5}$ (sugarcane); CH_4 (jute)	United States (laboratory based values)
[ze] Cao et al. (2008)	SO_2, OC (rice, wheat); CO, OC (maize); NO_x (millet, potato, oil crops); CO_2, CO, CH_4, $PM_{2.5}$ (jute)	China (laboratory based values)
[zm] Jayarathne et al. (2018)	$PM_{2.5}$ (oil crops, pulses)	Nepal (field experiment based value from, NAMaSTE campaign)

Suite (VIIRS) fire counts in agricultural land (Fire Information for Resource Management System, 2020) were considered (Ni et al., 2015).

2.6. *Emission uncertainties*

In this study, the Monte Carlo method was used to analyze the uncertainties of the emission inventory (e.g., Zhang et al., 2019; Zhou et al., 2017; Zhao et al., 2011; Ni et al., 2015). In a few studies, the coefficient of variation (CV) for activity data for straw burning has been assumed as 20% (Ni et al., 2015; Zhou et al., 2017). Following them, the uncertainty of residue burned was set at 20%. The major factor that determines the uncertainties of emissions is EFs (Zhang et al., 2019). Thus, in selecting the EFs, priority was given to country-based EFs. Only in cases where country values were not available, regional and global EFs were used. Regional and global EFs vary considerably in the literature. Hence, the mean and SD were calculated from the sets of available data (section 2.4, Supplementary Fig. 1). The range of residue burned, EFs and emissions were calculated by averaging 20,000 Monte Carlo simulations with a 95% confidence interval converted to a percentage of the mean (see Supplementary Table 9–11).

3. Results and discussion

3.1. *Trends in dry matter generation*

On average, the trends in dry matter (DM) generation have increased from 2003/04 to 2016/17 (see Supplementary Figure 3). It was estimated to be 13,500 Gg in 2016/17, 29% higher than the value for 2003/04. In 2016/17, the total DM generation from paddy stalk was 6670 Gg followed by wheat stalk (2340 Gg), maize stalk (1840 Gg), sugarcane residue (854 Gg), potato residue (606 Gg), pulses residue (460 Gg), oil crop residue (343 Gg), millet residue (294 Gg), barley residue (32.9 Gg) and jute residue (11.3 Gg) (see Supplementary Table 2).

3.2. *Trends in fuel use, animal fodder and roof thatching*

On average, trend in crop residue consumed as fuel has marked an increase (see Supplementary Table 3). It was estimated to be 1410 Gg in 2016/17, 29% higher than the value for 2003/04. Of the total dry matter produced, 76% was used as animal fodder. On average, the DM consumption trend has increased with the recorded in 2016/17 (i.e., 10,200 Gg). Nationally, the dry fodder consumption trend has shown an increase from 8820 Gg (2003/04) to 10,200 Gg (2016/17) (i.e., by 16%). Likewise, the trend in dry residue use for roof thatching has shown an increase from 114 Gg (2003/04) to 133 Gg (2016/17) (i.e., by 17%).

3.3. *Trends in crop residue open burning*

The amount of CROB was calculated by subtracting the total residue consumption pattern from total residue generation at district level. In districts where a residue consumption deficit prevailed, open burning was estimated to be zero. By adding up the surplus value after calculating the residue consumption pattern at each district, amount of residue to be openly burned was estimated. As Supplementary Figure 3 and Supplementary Table 4 show, the CROB trend has recorded an increase annually. The total CROB estimated as 2280 Gg in 2003/04 is seen to be slightly higher than the 2000 Gg for 1999/2000 reported by Streets et al. (2003). Similarly, it was estimated to be 2908 Gg (61–139%, i.e., uncertainty ranging in between 1770 and 4040 Gg) in 2016/17, which is 28% higher than that for 2003/04. But it shows a significant drop in 2015/16, which could be due to the severe earthquake that occurred in Nepal in April 2015. The regular agricultural practices of people would have been seriously disrupted by the extensive damage suffered during this emergency, in turn reducing CROB. In the year 2016/17, the fraction of residue burned in the field was estimated to be 25%. But the highest open burning in agricultural fields is recorded for 2013/14, with the fraction of residue burned calculated as 27%.

The fraction of CROB estimated from this study was compared with that given in the other published report on the subject in Nepal. The study by National Animal Feed and Livestock Quality Management Laboratory (2019) estimated that, in the Terai region of Nepal, 30% of the straw is openly burnt due to the use of the combine harvester. Streets et al. (2003) have in fact reported a default value of 25% for South Asia. Moreover, Sahai et al. (2010) and Irfan et al. (2014) have used the default value of 25% for India and Pakistan. The trends in open burning are associated with residue generation and residue consumption patterns (see Supplementary Figure 3).

3.4. *Trends in emissions*

Our study presents the national air pollutant emissions for the years 2003/04 to 2016/17. The results suggest an increase in the annual trend of emissions (see Fig. 1 & Table 4). It is noteworthy that the year 2013/14 has the highest emissions whereas the year 2006/07 records the lowest. In 2016/17, the greenhouse gases were estimated to be 4140 Gg for CO_2 (56–144%) and 6.5 Gg for CH_4 (7–193%). The other harmful pollutants were 154 for CO (4–196%), 1.2 for SO_2 (60–140%), 24.5 for $PM_{2.5}$ (30–170%), 8.6 for OC (38–162%), 2.2 for BC (-1-201%), 7 for NO_X (54–146%), 22.5 for NMVOC (8–192%) and 2.7 for NH_3 (3–197%) in unit of Gg (see Table 4, Supplementary Table 6, and Supplementary Table 7). The Tables show that pollutants such as, CO_2, CO, CH_4, OC, $PM_{2.5}$, BC, NOx, NMVOC, and NH_3 increased by 27%, a substantial increase from 2003/04 to 2016/17. During the time-period, the highest emissions were recorded in 2013/14 whereas the lowest were recorded in 2006/07. Supplementary Table 5 presents the emissions of pollutants district-wise.

The emissions from this study were compared with the technology-based emission inventory for 2011 for Nepal (Sadavarte et al., 2019). Our results suggest that CO_2 emissions were 2.1, CO 11.1, CH_4 16.9, SO_2 20, OC 9.7, $PM_{2.5}$ 8, BC 10.5, NO_X 9.1, and NMVOC 18.1 folds lower respectively than the reported literature (see Supplementary Table 6). The emissions were also compared with those from China (Ni et al., 2015; Li et al., 2017; Zhou et al., 2017; Zhang et al., 2019), India (Pandey et al., 2014) and South East Asia as well as with those for Central Asia in order to place these values in a global context (GFED, 2020). The comparison showed that emissions contribution for Nepal is significantly lower than that for China and India. The share of CO_2 with respect to South East Asia (GFED, 2020) was 10.35%; CO 5.92%; CH_4 4.33%; SO_2 12%; OC 17.2%; $PM_{2.5}$ 15.31%; BC 11.58%; NO_X 8.75%; and NH_3 4.82%. Similarly, the share of CO_2 with respect to the global context (GFED, 2020) was 1.15%, CO 0.65%; CH_4 0.48%; SO_2 1.29%; OC 1.62%; $PM_{2.5}$ 1.69%; BC 1.26%; NO_X 0.97%; and NH_3 0.54% (see Supplementary Table 7).

3.5. *Spatio-temporal variations and BC flux concentration*

Open burning varied according to the geographical region of the country. For the year 2016/17, the highest residue burning occurred in the Terai region (90.7%) followed by the hills (6.4%) and the mountains (2.9%). This could be due to various factors. The Terai region carried a higher capacity of agricultural production than the other two regions (hence, 66.8% of residue generation too) due to

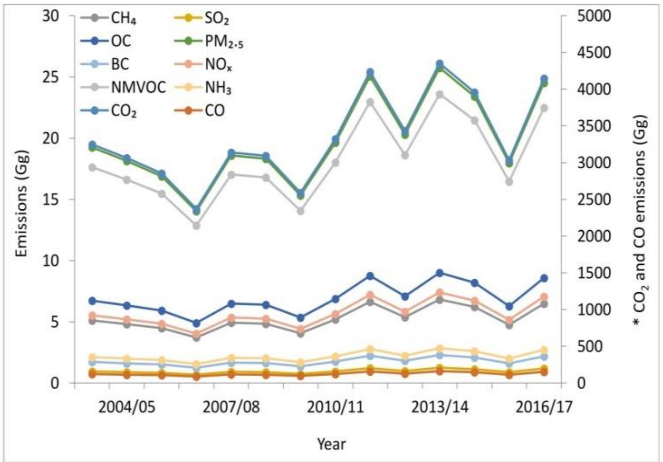

Fig. 1. National trends in emissions from crop residue open burning (2003–2017).

Table 4
National emissions trend from crop residue burning (in Gg).

Year	CO_2	CO	CH_4	SO_2	OC	$PM_{2.5}$	BC	NO_x	NMVOC	NH_3
2003/04	3250	120	5.1	0.9	6.7	19.2	1.7	5.5	17.6	2.1
2004/05	3060	114	4.8	0.9	6.3	18.1	1.6	5.2	16.6	2.0
2005/06	2850	106	4.5	0.8	5.9	16.9	1.5	4.9	15.5	1.9
2006/07	2370	88	3.7	0.7	4.9	14.0	1.2	4.0	12.9	1.5
2007/08	3140	116	4.9	0.9	6.5	18.6	1.6	5.3	17.0	2.0
2008/09	3090	115	4.9	0.9	6.4	18.3	1.6	5.3	16.8	2.0
2009/10	2590	96	4.1	0.7	5.4	15.3	1.4	4.4	14.0	1.7
2010/11	3320	123	5.2	1.0	6.9	19.6	1.7	5.6	18.0	2.2
2011/12	4240	157	6.6	1.2	8.8	25.1	2.2	7.2	23.0	2.8
2012/13	3430	127	5.4	1.0	7.1	20.3	1.8	5.8	18.6	2.2
2013/14	4350	161	6.8	1.3	9.0	25.7	2.3	7.4	23.6	2.8
2014/15	3960	147	6.2	1.1	8.2	23.4	2.1	6.7	21.5	2.6
2015/16	3030	112	4.8	0.9	6.3	18.0	1.6	5.2	16.5	2.0
2016/17	4140	154	6.5	1.2	8.6	24.5	2.2	7.0	22.5	2.7
Avg.	3340	124	5.2	1.0	6.9	19.8	1.8	5.7	18.1	2.2

the accessible land-use area, followed by the hills (27.7%) and the mountains (5.5%) (see Supplementary Fig. 2). In addition, both the supply of and demand for dry fodder consumption determined open burning from crop residue. Thus, in districts where there were feed deficit, open burning was low to none at all. The emissions were also low to zero (i.e., the mountain and hills). Likewise, in districts where feed were surplus, open burning was high with more emissions (see Fig. 2 & Supplementary Table 5). Studies have also suggested that the higher deployment of the combine harvesters in recent times has led to increased residue burning (Yang et al., 2008; Val-Aguasca et al., 2019; Gupta, 2014). In Nepal, there has been an increase in the usage of combine harvesters since 2000, especially in the Terai region (National Animal Feed and Livestock Quality Management Laboratory , 2019; Takeshima, 2017; Mandal et al., 2017). Increase in mechanization could be due to the paddy-wheat system (Gupta et al., 2004) as well as labor migration. This study thus explored also the linkage between open burning and national GDP/remittance due to labor migration (The World Bank, 2019) from 2003 to 2017. The graph shows a moderate correlation ($R^2 = 0.3002$) (see Supplementary Fig. 4). But the available literature suggests the combination of shortage of labor and high wages could lead to more open burning (Jethva et al., 2019; Gupta, 2014; Gupta et al., 2004). Studies have also reported that farmers intentionally burn residue for weeds and disease control as well as nutrient release for a better crop yield during the next cycle (Gadde et al., 2009; Chang et al., 2013; Kadam et al., 2000).

Fig. 2 gives the district-wise BC emissions (Grid 1 × 1 km) for the entire country under cultivated areas at the district level. The figure shows that while BC emission was widespread in the Terai region, it was the lowest in the mountains and hills (Fig. 2). BC is a high priority pollutant in Nepal and the Hindu Kush Himalayan region. It is the second most anthropogenic climate-forcer at present after carbon dioxide and is triggered by incomplete combustion (Bond et al., 2013). It plays a big role in the melting of Hindu Kush Himalayan glaciers (Yang et al., 2020; Yi et al., 2019; Bond et al., 2013). Moreover, fine BC has a significant impact on health (Highwood and Kinnersley, 2006).

Active fire counts were combined for the calculation of the temporal variations in emissions. The emissions were highly concentrated from February to May (with 86.16% of the total emissions), with the peak coming in April (see Fig. 3 & Supplementary Table 8). The open burning of crop residue of winter crops during this time could be the reason for the high air pollutants. However, with the onset of the regular monsoonal rains in Nepal, the agricultural residue open burning practices decline or do not occur in sizes and volumes that can be detected by a satellite. Therefore, emissions were estimated to be zero during this season.

3.6. *Air quality management opportunity*

The Terai region is exposed to more pollutants from agricultural residue open burning than the hills and mountains. Harmful pollutants from open burning of crop residue can trigger severe health impacts like respiratory disease, cardiovascular disease, allergies and premature deaths (Zhang et al., 2017) in addition to the adverse impact on climate as CO_2, CH_4, N_2O and BC have proven global warming potential (Li et al., 2017; Das et al., 2018). According to Gupta (2014), open burning of residue is less likely to occur where farmers have more livestock per hectare. So raising livestock by local farmers for additional income could be potential mitigation measures to reduce open burning. The study conducted by Gupta

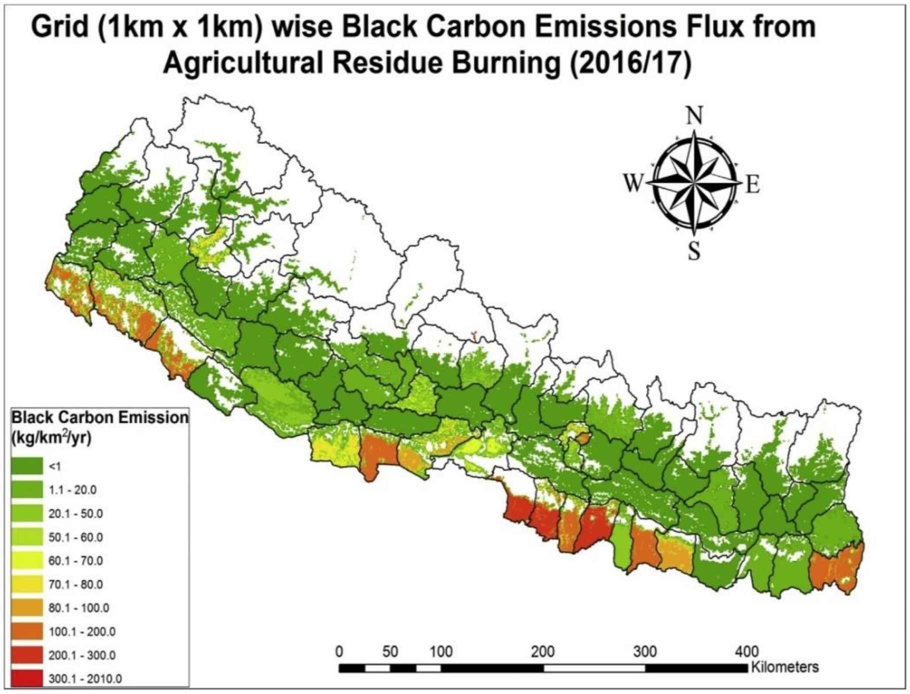

Fig. 2. BC emission flux from crop residue open burning for year 2016/17.

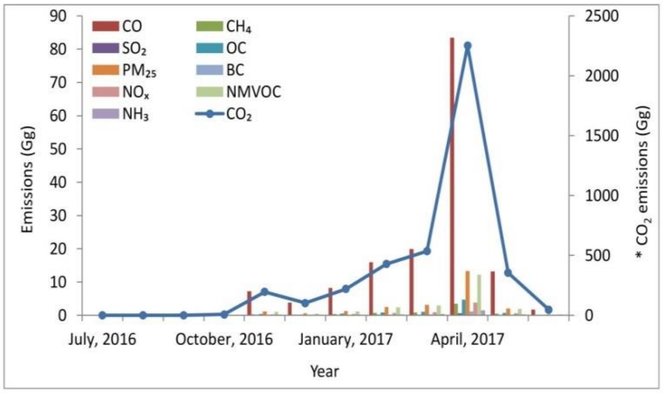

Fig. 3. Monthly variation of emissions (2016/17).

(2014) and Gupta et al. (2004) suggest combine harvesters can be modified to collect crop residue separately while harvesting. The chopped residue can be used for animal feed, alternative energy production (e.g., bio-briquette) and raw materials for industries (e.g., mushroom cultivation, paper production, brick kilns, etc.). Reaper harvesting technology can also be deployed to collect the left residue from the field. For commercial farming, combine harvester can be fitted with residue spreader to distribute loose crop residue uniformly to plant wheat through happy seeder technology (Gupta, 2014). The analysis of temporal distribution from open burning of crop residue showed that emissions were not evenly throughout the year. Around 54% of the total emissions occurred in the month of April and zero from July to September (see Fig. 3). This is to be noted and should be considered by policy makers for better air quality management.

4. Conclusions

The study estimates the national air pollutant emissions for the years 2003/04 to 2016/17, with 2013/14 showing the highest emissions. The total emissions from CROB in Nepal for 2016/17 were estimated to be 4140 of CO_2 (56–144%), 154 of CO (4–196%), 6.5 of CH_4 (7–193%), 1.2 of SO_2 (60–140%), 24.5 of $PM_{2.5}$ (30–170%), 8.6 of OC (38–162%), 2.2 of BC (-1-201%), 7 of NO_x (54–146%), 22.5 of NMVOC (8–192%) and 2.7 of NH_3 (3–197%) in unit of Gg yr^{-1}. February to May accounted for 86.16% of the total emissions, with the peak occurring in April. However, the results indicate that no emissions occurred between July and September. It is highlighted here that the CO_2 emission considered in this study is from the combustion in the agricultural field and doesn't represent net emissions. To estimate the net CO_2 emission, various factors need to be considered which is beyond the scope of the present study but could be undertaken in a future study. Based on the estimates, a high resolution (1 km × 1 km) model-ready emissions inventory was developed for Nepal to better understand the spatial distributions of air pollutants from open burning for 2016/17. The study calculated the emissions flux of BC, which suggests the Terai region to be the highest emitter of BC followed by the hills and mountains. The main factor behind open burning was residue generation and its consumption pattern. The districts where residue consumption demand exceeded the generation demand, open burning was estimated to be zero, which was also closely associated with livestock rearing. This study also calculated the correlation between National GDP due to remittance/labor migration and open burning from 2003 to 2017, which showed a moderate correlation ($R^2 = 0.3002$). The study found the air quality due to CROB to be more vulnerable in the Terai region (Southern Nepal) than in the hills and mountain. Therefore, alternative uses of residue (for animal feed, bio-briquette production, raw materials for industries, etc.), deployment of modern technology (modified combine harvesters, reaper harvester, and happy seeder) and emission control policies ought to be considered among the foremost steps to reduce air pollution.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2020.115069.

The emission flux is highlighted by ten different colors. The lowest emission flux to highest emission flux is indicated by dark green to dark red. The white region in each district represents an area where the emission flux was zero. The dark green region is an area where residue burning is estimated to be insignificant.

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Estimating emissions from open burning of municipal solid waste in municipalities of Nepal

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ABSTRACT

Open burning of municipal solid waste (MSW) is a poorly-characterized and frequently-underestimated source of air pollution in developing countries. This paper estimates the quantity of MSW that was burned in five erstwhile municipalities of the Kathmandu valley, Nepal. A household survey, a transect walk survey, an experiment to measure the fraction of waste that is combustible, a survey on fraction of population burning waste outside their houses, and a survey of the fraction of MSW burned at dump sites were performed in this study, whereas burning/oxidation efficiency, municipal populations, MSW generation rates, and emission factors were derived from the literature. The total mass of MSW burned during 2016 is estimated to be 7400 tons (i.e., 20 tons/day), which was of 3% of the total MSW generated in the valley municipalities that year. This exceeds Government estimates by a factor of three. Multiplying the burned MSW mass by emission factors, the air pollutant emissions are estimated as $PM_{2.5}$ 55 tons (OC 42 tons and EC 1.4 tons), PM_{10} 60 tons, BC 25 tons, CO_2 11,900 tons, CH_4 30 tons, SO_2 5.0 tons, NO_x 19.2 tons, CO 630 tons, NMVOC 112 tons, and NH_3 5.7 tons per year. Open burning of MSW can trigger health impacts such as acute and chronic respiratory disease, heart diseases, and allergic hypersensitivity, in addition to impacts on local climate. Improved waste-segregation practices at the source and waste-collection systems throughout the valley are needed to mitigate this pollution source and its effects.

1. Introduction

Solid waste management (SWM) has become a major concern, especially in urban areas of developing countries. Many municipalities are experiencing extreme environmental degradation as well as public health risks due to ill-timed waste management and unsanitary disposal practices (Alam et al., 2008; Nagpure et al., 2015). Recently, the open burning of solid waste was implicated as a major cause for soiling the Taj Mahal and impairing the health of Agra residents (Lal et al., 2016). In Nepal, population growth, rapid expansion of sprawling urban municipalities, increasing amounts of industrial and commercial activity, and rising consumption of packaged goods has resulted in severe air and water quality issues, poor sanitation, and the spread of diseases (Alam et al., 2008; Dangi, 2009; Pokhrel and Vivaraghavan, 2005).

At an elevation of 1400 m, the bowl-shaped Kathmandu valley lies at the foothills of the Himalayas and is surrounded by mountains and forests. The total urban area of Kathmandu valley is 96.68 km^2 (KVDA, 2017) and this area has the highest population density in Nepal. The valley contains five densely-inhabited urban centres which were previously designated as municipalities: Kathmandu Metropolitan City (KMC), Lalitpur Sub-Metropolitan City (LSMC), Bhaktapur, Kirtipur and Madhyapur Thimi. Around the time of study, the Government of Nepal designated 16 municipalities (dividing many of the earlier five into smaller areas) in the valley partly in response to the booming urban population (KVDA, 2017).

Fig. 1 contains a map of the five original municipalities referred to throughout this study and their location within Nepal. KMC is home to the nation's capital and is the most populated municipality in Nepal with an area of 49.45 km^2 (CBS, 2013) subdivided into 35 wards (KMC, 2014). LSMC was the country's third most populous municipality and is located in the south-central part of the valley, covering an area of 24.94 km^2 that was subdivided into 30 wards (LSMC, 2016). Bhaktapur is an ancient city in the eastern

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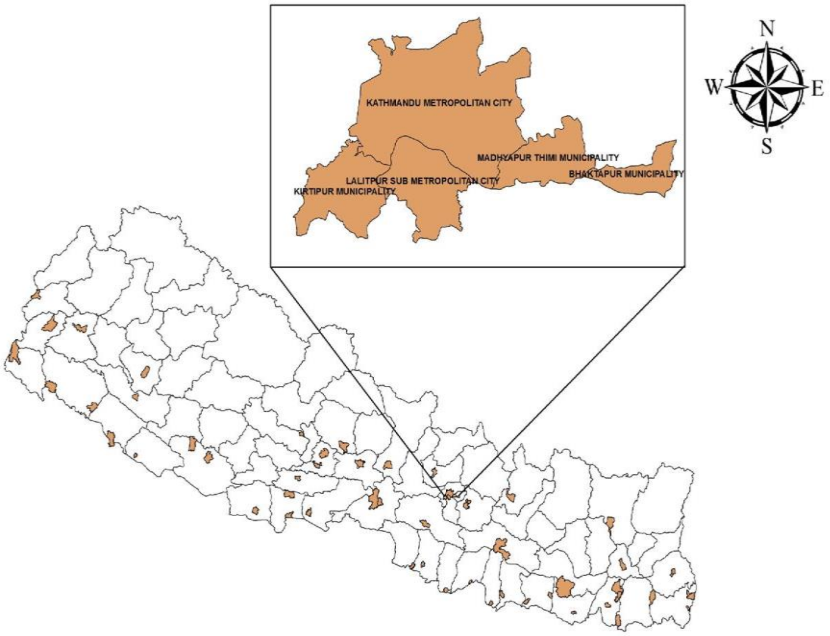

Fig. 1. Map of Nepal with political boundaries delineating the 75 districts (mostly white) and the 58 old municipalities (orange). The enlarged portion shows the five Kathmandu valley municipalities where this study focused. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

part of the valley, 16 km east of the KMC centre. It was divided into 17 wards and covered only 6.88 km^2 (Bhaktapur Municipality, 2016). Kirtipur municipality is less populated than the others and has grown around another ancient kingdom southwest of the KMC centre. Madhyapur Thimi is the newest of the five municipalities, which resulted from population infill between Bhaktapur and KMC.

Although Kathmandu valley is the most developed place in Nepal and covers the largest number of commercial and institutional sectors, the SWM in much of the valley remains unsatisfactory. In Kathmandu, SWM has become a chronic problem that has challenged and evaded development efforts for decades (Dangi, 2009). Many statistics related to SWM in Kathmandu valley are highly uncertain, as evidenced by the large variability across studies (Nippon Koei Co. Ltd. and Yachiyo Engineering Co. Ltd., 2005; SWMRMC, 2008; Dangi et al., 2011). A brief summary is provided here with the acknowledgement that these data are not necessarily consistent with each other.

About 50% of the waste from municipalities of Kathmandu valley is generated by households, 43% from commercial, 6% from the institutional, and relatively little (1%) from parks and gardens, street sweeping, and from neighboring villages (ADB, 2013). In 2012, the average per capita municipal solid waste generation rates (MSWGR) for KMC, LSMC, Kirtipur, Bhaktapur and Madhyapur Thimi were 0.46 kg/capita/day (kcd), 0.37 kcd, 0.25 kcd, 0.35 kcd, and 0.27 kcd, respectively. The most recent study (SWMTSC, 2015) showed the average MSWGR for a sub-urban neighborhood of KMC (i.e., Budanilkantha) and LSMC (i.e., Mahalaxmi/Gwarko) in 2014 to be 0.48 kcd and 0.36 kcd, respectively. Comparing ADB (2013) to the past studies (Nippon Koei Co. Ltd. and Yachiyo Engineering Co. Ltd., 2005; Alam et al., 2008), the per capita MSWGR has increased steadily in the valley municipalities. This is likely due to increased consumption of packaged goods and gradual rise of commercial and industrial activities. Moreover, a research-grade study by Dangi et al. (2011) indicates that the MSWGR is even higher than all of the aforementioned reports.

Although 71% of MSW generated in Kathmandu is organic (Dangi et al., 2011), very few neighborhoods have systems in place to compost this material (Sherpa, 2017). Within the valley municipalities, only 35.3% of waste from Kirtipur, Madhyapur Thimi (52.2%), LSMC (71.2%), Bhaktapur (86.5%) and KMC (86.9%) are collected (ADB, 2013). The situation in other low-income countries appears to be similarly abysmal despite the sizeable expenditure of financial resources on SWM (World Bank, 2012; Hazra and Goel, 2009). At present, urban areas receive more attention for MSW open burning because they are highly populated (Wang et al., 2017).

Large quantities of uncollected waste can be found along the banks of urban waterways such as Bagmati and Bishnumati (Pokhrel and Vivaraghavan, 2005). The water from these rivers is used for domestic purpose, cultivating agriculture and also has religious significance for Hindus. Sometimes uncollected waste is found in close proximity to small-scale agricultural fields where it contaminates the food supply. In communities that are far from waste-collection routes (Subedi, 2016), refuse is commonly dumped in privately-owned lots that are neither developed nor maintained (Bajracharya, 2016). Any waste that remains uncollected after a few weeks of biodegradation emits a foul odor, prompting nearby residents to burn it (Bajracharya, 2016; Sherchan, 2016).

At this point, it is useful to note that the primary issue discussed in this paper is from MSW that is burned in the open, where combustion conditions (i.e., low temperature, suboptimal air-to-fuel ratio, high moisture content) are favorable to pollutant formation (Wiedinmyer et al., 2014). This paper provides no commentary about the high-temperature, carefully-controlled incineration that is practiced in some developed countries. Open burning of MSW has a major impact on human health because the emitted smoke contains life-threatening particulate matter (PM) enriched in organic carbon (OC) and elemental carbon (EC), carcinogenic dioxins, and numerous other harmful pollutants like nitrogen oxides (NO_x), sulfur dioxide (SO_2), carbon monoxide (CO), non-methane volatile organic compound (NMVOC) (Nagpure et al., 2015; Guttikunda et al., 2014; Guttikunda, 2007; Hodzic et al., 2012; Wiedinmyer et al., 2014; Pokhrel and Vivaraghavan, 2005). High exposure to PM may lead to respiratory and cardiovascular disease, cancer and adverse birth outcomes (McDonnell et al., 2000). It also exacerbates local/regional warming because the carbon dioxide (CO_2) and black carbon (BC) emitted from burning both have a high global warming potential. Even if it is dumped and never burned, the gradual degradation of MSW emits methane (CH_4) which is another potent greenhouse gas. MSW open burning also emits ammonia (NH_3) which can enhance the formation of PM in the atmosphere.

Unlike vehicle exhaust and industrial emissions, the common public and elected officials are relatively unaware of the significance and harmful impacts of open burning (Sherchan, 2016). In the few emission inventories that have been made, open burning of MSW is either completely neglected (Pradhan et al., 2012) or very crudely estimated (Wiedinmyer et al., 2014; Gautam, 2006). This is mainly because of the difficulties and large uncertainty in estimating how much MSW is actually burned (Nagpure et al., 2015). The only urban-scale estimate of emissions from open burning of MSW in Kathmandu was made a decade ago and it arbitrarily assumed that 1% of the waste generated is burned (Gautam, 2006). Other sources of uncertainties are: (i) emissions factors; (ii) activity parameters such as the fraction of population that burns their waste and the fraction of waste that is combustible; and (iii) spatial allocation of waste-burning activities. The purpose of the present study is to constrain some of these uncertainties so that the importance of MSW burning relative to other air pollution sources in Kathmandu Valley may be better understood.

2. Data and methods

This study estimates the MSW open burning in selected urban and sub-urban neighborhoods of Kathmandu valley, following a method that was successful in Delhi and Agra (Nagpure et al., 2015).

2.1. Preparation of field surveyors

Surveyors selected for this study received an orientation from the project scientists. A pilot-scale survey was conducted and repeated until the surveyors were confident and skilled enough to collect their field data independently.

A field study was conducted from April to June 2016 in seven selected portions of KMC including Budanilkantha, LSMC, and Bhaktapur municipality. The study areas included Kalimati/Dallu (core) and Baneshwor (core) of KMC; Budanilkantha (sub-urban); Lagankhel (core) and Mahalaxmi/Gwarko (sub-urban) of LSMC; and Sukuldhoka/Durbar square (core) and Kamalbinayak/Chyamasingha (sub-urban) of Bhaktapur municipality. Within these seven study areas, household surveys and transect walks were conducted along a total of 19 different routes (Fig. 2).

2.2. Estimation of the waste combustible fraction

The field based experiment on waste combustible fraction (δ) was carried out using five samples of MSW found in Kirtipur near Tribhuvan University. The samples were the mixture of household and commercial waste. First, the composition of each pile was noted by visual inspection. Second, initial mass of MSW was measured using a digital weighing balance. Third, the volume of each MSW pile was determined using a measuring tape and stick (e.g., length, width and height). Fourth, the different waste piles were ignited and carefully monitored until the combustible material had turned to ash. Fifth, the volume of each pile after combustion was measured as described above. Lastly, the burnt trash (ash) and the residual trash were segregated carefully and their masses were measured separately.

The burnt mass of MSW was calculated by subtracting the mass of residual trash left after combustion from the total initial mass. The ratio of burnt mass of MSW to total initial mass of MSW is referred to as the waste combustible fraction (IPCC, 2006).

$$\delta = \frac{\text{Burnt Mass of Trash}}{\text{Total initial mass of Trash}} \tag{1}$$

This experimentally-derived parameter was used in the subsequent calculations of MSW open burning.

2.3. Transect walks

In each of the seven study areas, the available major pathways to access the market centre were selected as routes for the transect walks. A distance-sampling approach (i.e., line transect method) was adopted (Nagpure et al., 2015) along each of the 19 routes. Each time a pile of waste was encountered along a transect, the surveying team recorded the geographical coordinates of that pile, and measured the dimensions (i.e., length, breadth and height) of the entire pile so that the total volume could be estimated. The study team took a digital photograph of each waste pile and recorded the time at which it was observed. A few days later, the identical transect walk was repeated a second time to determine whether some of the unburned waste piles were being collected or burned to ash and if new waste piles were introduced. The total number of waste piles was the sum of initial waste piles number from the first transect walk and new waste piles found during the second transect walk.

2.4. Household survey

While traversing each study route and observing waste piles, household surveys were also conducted. Thus, the sample size was determined based on the number of routes, household density and the size/length of each routes of urban core and sub-urban area. On routes with a sparse settlement pattern, every fifth household was selected for the survey starting with the first household along each route (i.e., 1, 6, 11, etc.). Along more densely-populated routes, approximately every tenth household was surveyed (i.e., 1, 11, 21, etc.). In this manner, a total of 179 households were surveyed (29, 35, 28, 16, 25, 23 and 23 in Kalimati/Dallu, Baneshwor, Budanilkantha, Lagankhel, Mahalaxmi/Gwarko, Bhaktapur core, and Bhaktapur sub urban respectively). Residents reported the number of occupants in their household, frequency of waste collection from their home, and their most common waste-disposal practices (e.g., dump, burn, compost, and/or give to collector).

2.5. Estimation of the fraction of population burning MSW

During transect walk, the total number of waste piles and burning incidence along the study routes were identified and noted down. Simultaneously, household surveys were conducted to know the status of total population residing in waste piles burning route and whether they were participating to the waste collection

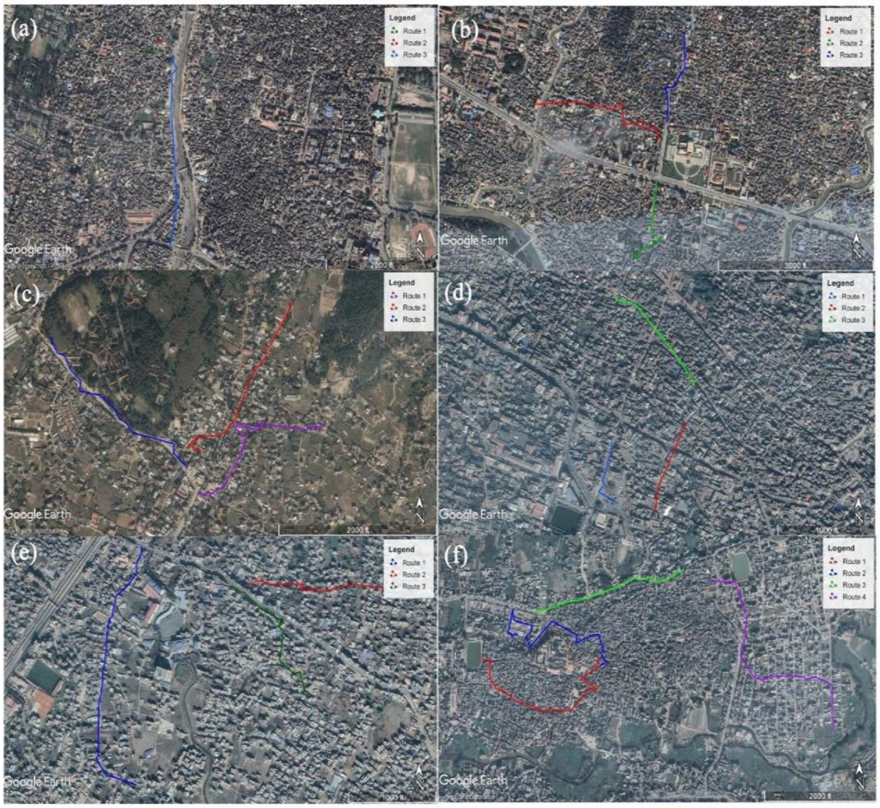

Fig. 2. Study routes. (a) Kalimati/Dallu (urban core); (b) Baneshwor (urban core); (c) Budanilkantha (sub-urban); (d) Lagankhel (urban core); (e) Mahalaxmi/Gwarko (sub-urban); (f) Bhaktapur (urban core and sub-urban).

services. Those who were not participating to waste collection services, the information of their methods of waste management practices (e.g., bury, burning, and composting) were obtained while interviewing. The household perception data of waste burning was then tabulated. Fraction of population burning MSW (P_{frac}) was estimated, which the ratio is of population who is not participating at waste collection to population whose waste is collected for disposal or landfilling (IPCC, 2006). From the above statement, population who was not participating at waste collection services is assumed to burn their waste. The P_{frac} was correlated with observation-based waste piles burning incidence through transect walk (i.e., waste piles burning per household).

2.6. Estimation of the fraction of MSW burning at disposal sites

To know the status of waste that was really dumped or burnt in the disposal site (λ), waste collection vehicles were tracked from waste collection points to disposal points in all the study routes. Study team visited to each waste collection points before 5:00 in the morning. To track the vehicles, motorbikes were used. Before heading to this study, different stakeholders (i.e., Department of Environment, KMC, Teku; Department of Environment, LSMC, Balkumari; Bhaktapur municipality; and locals) were consulted about the waste collection vehicle types, time, routes, frequency of collection, and availability of drivers. After the vehicles have been followed, visual inspection was made thoroughly at the disposal sites to check whether there were any signs of MSW open burning practices. Moreover, digital photographs were taken for visual estimation in the data analysis later.

2.7. Municipal solid waste generation

The series of study on MSWGR are not conducted in the municipality level in Nepal. The most widely used studies are reported to be SWMTSC (2015), ADB (2013), Dangi et al. (2011), SWMRMC (2008), Alam et al. (2008), and Nippon Koei Co. Ltd. and Yachiyo Engineering Co. Ltd. (2005). For this study, ADB (2013) and SWMTSC (2015) were considered because those are the latest studies and contain information of all municipalities which were studied.

2.8. Emission estimation method

The emissions from MSW open burning of valley municipalities were calculated. To calculate MSW burned at source and disposal sites of the urban and sub-urban neighborhood of valley municipalities, guidelines from different literatures were used (e.g., Shrestha et al., 2013a,b; IPCC, 2006; Shrestha, 2014).

2.8.1. Solid waste open burning at source

$$M_s = Pc \times MSWGR \times \delta \times P_{frac} \times \eta \times 365 \quad (2)$$

where M_s is the amount of open-burned MSW (kg/year); Pc is population (capita); MSWGR is per capita MSW generation rate (kcd); δ is fraction of combustible MSW; P_{frac} is fraction of population burning waste; and η is burning/oxidation efficiency (fraction), which is 0.4 (compiled by Shrestha, 2014).

2.8.2. Solid waste open burning at disposal site

$$M_s = Pc \times MSWGR \times \varepsilon \times \lambda \times \delta \times \eta \times 365 \quad (3)$$

where M_s is amount of open-burned MSW (kg/yr); Pc is population (capita); MSWGR is per capita MSW generation rate (kcd); ε is MSW collection efficiency (fraction that is disposed/land filled); λ is fraction of the waste that is actually burned relative to the total amount of waste disposed at a disposal site; δ is fraction of combustible MSW, and η is burning/oxidation efficiency (fraction).

2.8.3. Emissions

The total emission from MSW open burning can be estimated by multiplying activity data with emission factors (Shrestha et al., 2013a,b; IPCC, 2006; Shrestha, 2014; Defra, 2009). Generally, emission factor is expressed as grams of pollutant emitted per kilogram of fuel burned. The activity data (e.g., MSW generations) are the baseline for the emission estimation. The emissions for valley municipalities were estimated using equation (4) as reported by the above literatures.

$$Em_i = M_s \times EF_i \quad (4)$$

where Em_i is emission of pollutant i; EF_i is emission factor of pollutant i; M_s is amount of MSW burned.

The emission factors (EFs) presented in this paper is obtained from various literatures (i.e., country's specific as well as global based measurement). For estimating the mass of MSW open burning, EFs for CO, CO_2, NO_x, BC, $PM_{2.5}$, CH_4, OC, NH_3, and EC were obtained from the recently on-field measurements during NAMASTE campaign in Nepal (Stockwell et al., 2016; Jayarathne et al., 2018). Because of lack of country specific EFs for SO_2, global EF was considered which is a laboratory based measurement (Akagi et al., 2011). Moreover, EFs for PM_{10} and NMVOC were selected which is also a global based measurement (USEPA, 1995).

2.9. Data analysis and presentation

The computational tool such as MS-Excel was used to compile and analyse the data, whereas, R-Studio and Google earth were used to present the results. R-Studio is free and open-source software, which is used for data analysis, developing various graphics and modelling the findings in the field of environment, ecology, soil science and other scientific disciplines. In this study, R-studio is used for outlining per capita waste burnt accordingly geographic location of each study sites. Moreover, Google earth is used for delineating study routes and waste burning points. Arc-GIS is also used for presenting the study map.

3. Results and discussion

3.1. Population demography

The numbers of households counted along the study routes are shown in Table 3. Their population is determined using the average household size calculated from the survey results. The data of urban population for 2011 and projected population for 2016 for Nepal were 4,523,821 and 5,552,712 respectively (CBS, 2014a,b). The total population of 2011 for KMC, LSMC, Kirtipur, Bhakatapur and Madhyapur Thimi was 1,426,641 (CBS, 2012). Based on the above information, the projected population for 2016 for valley municipalities is estimated to be 1,751,114.

3.2. Waste combustible fraction

The waste combustible fraction was determined experimentally (see Section 2.2). Values ranged from 0.53 to 0.66 with an average of 0.57 (Fig. 3). From visual inspection, the waste piles were estimated to be 50% organic, 30% plastic, 10% paper, 3% rubbers, 2% metals, 2% textile, 1% glass and 2% other material. Moreover, sample organic waste piles were wet, whereas rest fractions were dry. Waste combustible fraction experiment was carried out immediately after rain disappeared. Thus, the results were based on all the aforementioned factors.

The experimental value presents the real-world emission parameter of Kathmandu valley municipalities, which is not adopted by any national study yet. As a pilot study, the experiment was limited to one site and five samples. As all five municipalities lie together which shares the same geographic boundary and similar socio-economic activities, lifestyles and climatic condition, the waste composition of each municipality is not expected to vary greatly. Therefore, it is reasonable to assume that the average waste combustible fraction value (0.57) determined in Kirtipur can be applied to all valley municipalities.

3.3. Estimation of the fraction of population burning MSW

To calculate MSW open burning emissions, fraction of population burning MSW is fundamental. The P_{frac} was calculated for

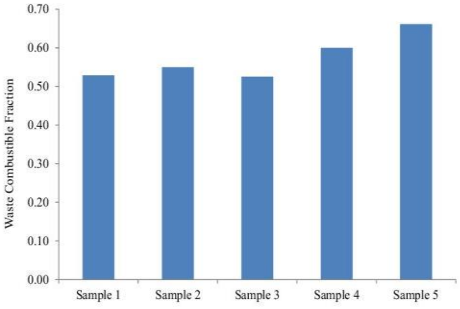

Fig. 3. Waste combustible fraction based on field experiment, 2016.

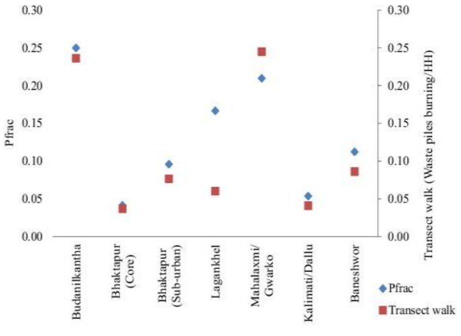

Fig. 4. Pfrac with respect to transect observation.

the valley municipalities referring the survey data. Of 179 households sampled, the average value of P_{frac} was 0.13 (Table 3). P_{frac} for Budanilkantha was comparatively higher than any other neighborhood. This could be due to irregular waste collection services because of less accessible, narrow and sloped roads to reach many of these homes. Higher value of P_{frac} infers higher waste burning or low waste collection efficiency in the neighborhoods. To validate P_{frac} with transect walk (i.e., waste piles burning per household), coefficient of correlation was calculated which is 0.88. P_{frac} has a strong correlation with MSW open burning along the transect line (Fig. 4). To some extent Lagankhel has a distinct case. In Lagankhel, some people burn waste in front of their houses and some burn at the nearest temporary informal dumping site (e.g., Lagankhel Bus Park). This could be the reason P_{frac} and MSW open burning incidence (e.g., waste piles burning per household) along transect of Lagankhel are feebly allied and coefficient of correlation to not exceed 0.88.

3.4. Estimation of the fraction of MSW open burning at disposal sites

All the waste that was collected through vehicles was tracked up to the final transfer station and disposal sites. The observations

Table 1
Total number of waste piles at the study routes.

Study routes	R1	R2	R3	R4	Total
Budanilkantha	19	20	21	–	60
Bhaktapur (core)	30	29	–	–	59
Bhaktapur (sub-urban)	–	–	25	24	49
Lagankhel	13	4	8	–	25
Mahalaxmi/Gwarko	28	21	24	–	73
Kalimati/Dallu	3	2	15	–	20
Baneshwor	11	11	11	–	33
Total					319

Table 2
Fraction of MSW open burning.

Study routes	Total no. of waste piles	No. of waste piles burning incidence	No. of waste piles non-burning incidence	Waste piles burning incidence (%)	Waste piles non-burning incidence (%)
Budanilkantha	60	39	21	65.00	35.00
Bhaktapur (core)	59	4	55	6.780	93.220
Bhaktapur (sub-urban)	49	17	32	34.69	65.31
Lagankhel	25	7	18	28.00	72.00
Mahalaxmi/Gwarko	73	49	24	67.12	32.88
Kalimati/Dallu	20	6	14	30.00	70.00
Baneshwor	33	15	18	45.45	54.55
In overall	319	137	182	42.95	57.05

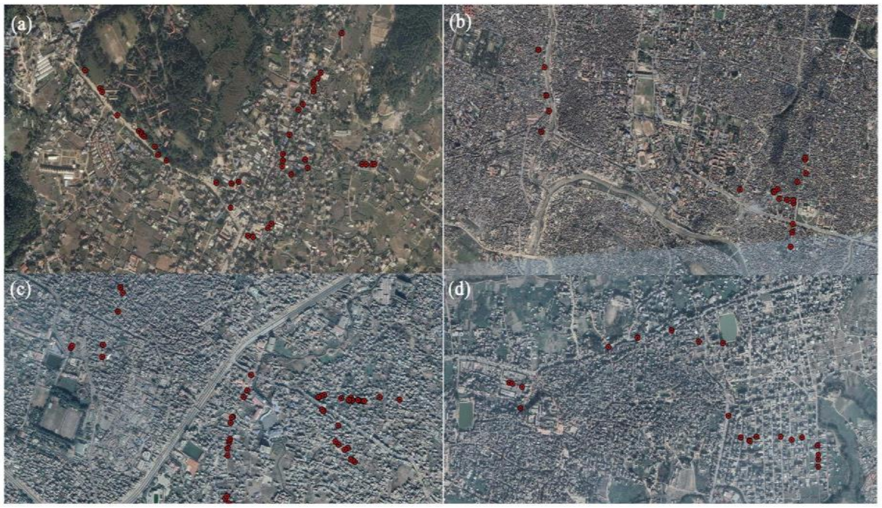

Fig. 5. Waste burning points at the study routes. (a) Budanilkantha (sub-urban); (b) Dallu/Kalimati (urban core) and Baneshwor (urban core) of KMC; (c) Mahalaxmi/Gwarko (sub-urban) and Lagankhel (urban core) of LSMC; (d) Bhaktapur (sub-urban and urban core).

have been made for each selected informal dumping site. The burning practices have not been identified in the disposal sites. The fraction of MSW-burned at the dumping sites is assumed as zero.

3.5. MSW open burning and emissions

Of 319 waste piles that have been identified (Tables 1 and 2), 137 (i.e., 43%) waste piles were found to be actively burning in different routes of the urban and sub-urban area (Fig. 5) of Kathmandu valley. Along transect of 1131 households study, the total mass of waste pile burnt is estimated to be 72 kg/day (Table 3). The highest waste burning was prevailing in Budanilkantha (0.027 kcd) followed by Mahalaxmi/Gwarko (0.017 kcd), Lagankhel (0.014 kcd), Baneshwor (0.012 kcd), Bhaktapur sub-urban (0.008 kcd), Kalimati/Dallu (0.006 kcd), and Bhaktapur core (0.003 kcd) (Fig. 6).

The household survey of Budanilkantha depicts a waste collection frequency to be very low (e.g., three times a week to twice a month). In contrast, Kalimati/Dallu and Baneshwor, which are the municipality core of KMC have low per capita waste burning because of higher waste collection efficiency (i.e., 86.9%), reported by ADB (2013). Mahalaxmi/Gwarko which is another sub-urban of LSMC has higher per capita waste burning and P_{frac} for the similar reason. The waste collection frequency as per survey is one to three times a week. In contrast, Lagankhel (i.e., municipality core of LSMC) has comparatively lower per capita waste burning, which closely relates with high waste collection efficiency (71.2%), reported by ADB (2013) and based on the survey (i.e., seven days a week). As compared to sub-urban neighborhoods, Bhaktapur core has lower per capita waste burning. This could be due to higher waste collection efficiency (i.e., 86.5%), reported by ADB (2013). To know the status of waste that was really dumped or burnt in the dumping site, vehicles were tracked by surveyors followed by visual inspection and digital photographs. In this way, the fraction of MSW burned at the disposal sites were determined as zero.

The total mass of MSW burned during 2016 for valley municipalities is estimated to be 7400 tons (i.e. 20 tons/day), which was of 3% of the total waste generated in that year. This result is compared with similar south Asian cities as well as other developing and developed countries. The mass of MSW burned is higher than other developed countries. European Union and United States determined MSW open burning rates around large cities, which are in between 0.25% and 0.3% of the total MSWGR (Park et al., 2013). The MSW burning of valley municipalities is 131,377, 197, 48, 11, 5, and 4 folds lower than global estimates, Mexico City, Mumbai, Delhi and Agra, Kanpur, and Ulaanbaatar respectively (Table 4). In Mexico City, MSW open burning is high (i.e., >50%)

Table 3
Summary of MSW open burning in the Kathmandu valley.

Study routes	Total no. of HHs around waste piles area	Avg. HH population size	Total no. of family members around waste piles area	Total weight of the trash that burnt (kg/day)	Daily per capita MSWGR (kg/capita/day)	Total daily MSW generation (kg/day)	P_{frac}	Waste burning (kg/capita/day)
Budanilkantha	165	4.86	801	21.86	0.48	383.15	0.25	0.027
Bhaktapur (core)	108	6.30	681	2.22	0.35	235.17	0.04	0.003
Bhaktapur (sub-urban)	222	5.43	1207	9.13	0.35	416.73	0.10	0.008
Lagankhel	116	4.50	522	7.38	0.37	194.09	0.17	0.014
Mahalaxmi/Gwarko	200	4.96	992	16.94	0.36	354.04	0.21	0.017
Kalimati/Dallu	146	5.14	750	4.27	0.46	348.52	0.05	0.006
Baneshwor	174	5.09	885	10.54	0.46	411.14	0.11	0.012
In overall	1131	5.16	5838	72.36	0.40	2342.85	0.13	0.012

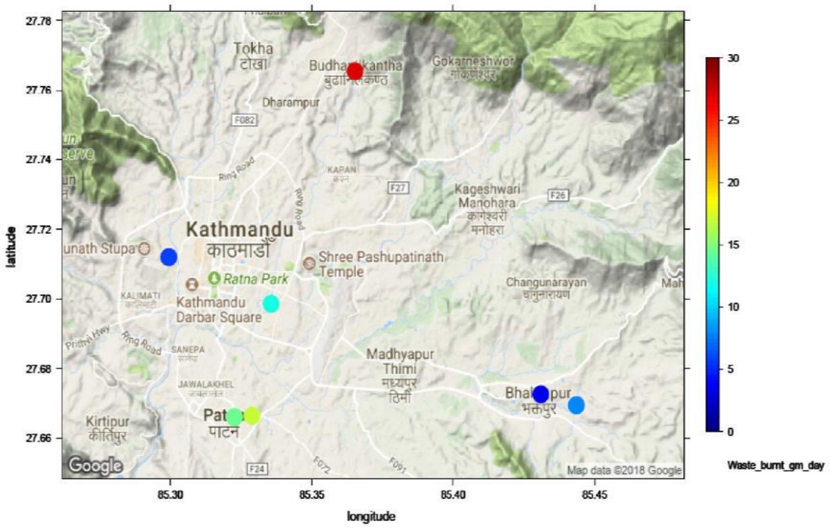

Fig. 6. Per capita waste burning (gm/capita/day) at the study sites.

Table 4
Comparative estimates of MSW open burning from global perspective.

Places	Waste burning (in Gg)
Kathmandu valley municipalities (2016)[a]	7.4
Delhi (2015)[b]	79.6
Agra (2015)[b]	81.4
Kanpur (2005)[c]	35.0
Mumbai (2005)[d]	352.6
Ulaanbaatar (2005)[e]	31.5
Mexico City (2006)[f]	1460.0
World (2014)[g]	972,190

MSW open burning percentage for 2005 is assumed same as that of 2007 for Kanpur.

[a] Estimation from this study.

[b] Nagpure et al. (2015).

[c] Calculated (MSW open burning percentage as of 2007 from Sharma, 2010; population as of 2005 from United Nations, 2012; MSWGR as of 2004/05 from CPCB, 2017).

[d] Calculated (MSW open burning percentage as of 2005 from Sharma, 2010; population as of 2005 from United Nations, 2012; MSWGR as of 2004/05 from CPCB, 2017).

[e] Calculated (MSW open burning percentage as of 2005 from Guttikunda, 2007; population as of 2005 from United Nations, 2012; MSWGR as of 2005, avg of summer and winter from JICA, 2007).

[f] Hodzic et al. (2012).

[g] Wiedinmyer et al. (2014).

in the poorest area (Hodzic et al., 2012). The global waste burning status is presented by Bond et al. (2004). According to it, Asia contributes 14 Tg/year, Africa (5 Tg/year) and global (33 Tg/year). This result is inconsistent with Wiedinmyer et al. (2014) which states MSW open burning to be 972 Tg/year after summing-up residential and dump sites burning. The waste burning varies from places and occurs due to variety of reasons. It is more frequent where waste collection services are sparse, expensive, and unavailable (Wiedinmyer et al., 2014). Small and medium level cities, and cities without landfill facilities and with no or partially waste collection service causes more MSW open burning (Guttikunda et al., 2014). Other reasons are lack of adequate waste collection services (Ramaswami et al., 2016) as well as irregular waste collection services, inept disposal methods, and poor attitude of dump-burn practices which lead to MSW open burning in residential and open spaces. Moreover, proper waste recycling systems are also trifling in the country. In average, the waste collection frequency in the Kathmandu valley is three times a week (SWMTSC, 2015).

This study estimates the total emissions from MSW open burning to be $PM_{2.5}$ 55 tons (OC 42 tons and EC 1.4 tons), PM_{10} 60 tons, BC 25 tons, CO_2 11,900 tons, CH_4 30 tons, SO_2 5.0 tons, NO_x 19.2 tons, CO 630 tons, NMVOC 112 tons, and NH_3 5.7 tons per year (Fig. 7). The results have been compared with existing estimates from Diesel Generator (DG) sets, vehicle traffic, and manufacturing industries of different years to establish MSW burning as a leading in the valley. World Bank (2014) reported DG sets emission for Kathmandu valley for 2012/2013. The estimates of SO_2, NO_x, CO, PM_{10}, BC, OC and CO_2 from MSW open burning are 11, 281, 2, 6, 9, 3, 18 folds lower than DG sets emissions respectively. Comparing waste burning emission with vehicular emission, NO_x is 834 folds lower, CO is 49 folds lower, BC is 86 folds lower and CO_2 is 130 folds lower than reported by Shrestha et al. (2013a,b). Comparing with manufacturing industries (Pradhan et al., 2012), MSW open burning emits SO_2 and NO_x which are 607 and 44 folds lower respectively. However, CO and NMVOC are 1.6 fold and 2 folds higher respectively than manufacturing industries (Table 5).

Moreover, emissions from this study are compared with other regional and international findings. The emissions such as SO_2, NO_x, CO, PM_{10}, and $PM_{2.5}$ resulted from trash burning from valley municipalities are 80, 104, 44, 128 and 97 folds lower than Delhi respectively. Similarly, SO_2, NO_x, CO, and NH_3 are 146, 209, 96

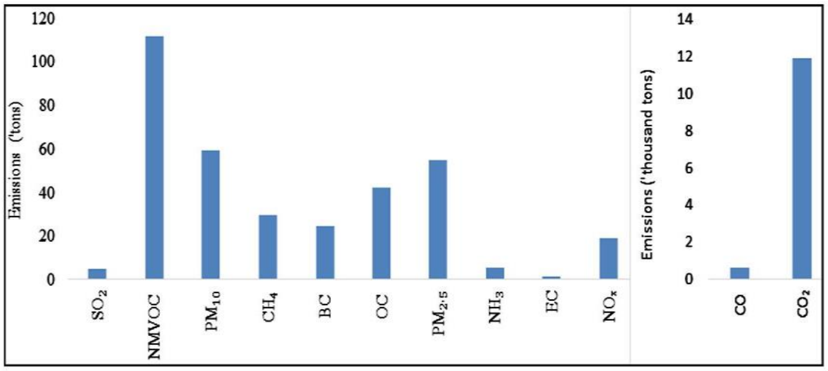

Fig. 7. Emissions from MSW open burning, valley municipalities.

Table 5
Comparative estimates of emissions from different sources in the Kathmandu valley.

Emission sources	SO_2	NO_x	CO	NMVOC	PM_{10}	CH_4	BC	OC	CO_2	$PM_{2.5}$	NH_3	EC
Solid waste burning – Kathmandu valley municipalities (2016)[a]	5	19.2	629.9	111.5	59.5	29.5	24.5	42.2	11,913	54.8	5.7	1.4
DG sets – Kathmandu valley (2012/13)[b]	54	5400	1200	–	380	–	221	120	210,150	–	–	–
Vehicular emission – Kathmandu valley (2010)[c]	–	16,000	31,000	–	–	–	2100	–	1,554,000	–	–	–
Manufacturing industries – Kathmandu valley (2012)[d]	3014	838	389	53	–	–	–	–	–	–	–	–

[a] The estimation from this study.

[b] World Bank (2014).

[c] Shrestha et al. (2013a,b).

[d] Pradhan et al. (2012).

Table 6
Comparative estimates of emissions (in Giga grams) from global perspective.

Emission sources	SO_2	NO_x	CO	NMVOC	PM_{10}	CH_4	BC	OC	CO_2	$PM_{2.5}$	NH_3	EC
Kathmandu valley municipalities (2016)[a]	0.01	0.02	0.63	0.11	0.06	0.03	0.02	0.04	11.91	0.05	0.01	0.00
Delhi (2010)[b]	0.40	2.00	27.80	–	7.60	–	–	–	–	5.30	–	–
Mexico city (2012)[c]	0.73	4.02	60.59	–	–	–	–	–	–	–	1.46	–
Ulaanbaatar (2006)[d]	–	–	–	–	4.07	–	–	–	–	3.05	–	–
World (2014)[e]	486	3636	36,943	–	11,569	3597	632	5123	1,412,592	9527	1089	–

[a] The estimation from this study.
[b] Guttikunda and Calori (2013).
[c] Hodzic et al. (2012).
[d] Guttikunda (2007).
[e] Wiedinmyer et al. (2014).

and 256 folds lower than Mexico City respectively. Furthermore, PM_{10} and $PM_{2.5}$ are 68 and 56 folds lower than Ulaanbaatar respectively. Likewise, SO_2, NO_x, CO, PM_{10}, CH_4, BC, OC, CO_2, $PM_{2.5}$ and NH_3 are 97,219, 189,375, 58,649, 194,438, 121,936, 25,793, 121,409, 118,576, 173,859, and 191,027 folds lower than global estimates respectively (Table 6). Some other study shows that MSW burning contributes direct PM emissions, approximately 8% in India, 22% in China, 33% in Bangladesh, and 69% in Pakistan (Wiedinmyer et al., 2014). In Mumbai, waste burning and landfill fires emit 22,000 tons per year of air pollution in the form of PM, HC, CO, NO_x, and SO_2 including dioxins/furans (10,000 TEQ grams annually) (Annepu, 2012). These emission can lead to severe health impact like acute and chronic respiratory disease, heart diseases, and allergic hypersensitivity (McDonnell et al., 2000) including impacts on local climate because BC and CO_2 having a high global warming potential.

4. Conclusion

In Nepal, this study for the first time attempted to estimate real-world emission parameters, which have been used for estimating MSW open burning. Moreover, it also accounts daily waste burning incidence in the Kathmandu valley. This study presents a clear picture that more MSW open burning prevails at the sub-urban neighborhoods than the municipality core. This is due to lower waste collection frequency (or efficiency) at sub-urban area than the municipality core. P_{frac} has strong linkage with waste collection efficiency. Higher the waste collection efficiency, lower will be P_{frac} and vice versa. This study reveals that open burning practices are not performed in the municipal designated disposal site.

The method applied in this study can be also replicated to more neighboring cities of the developing countries to obtain amount of pollutants from MSW open burning. For the cities of developing countries, which waste composition and climatic condition are similar to valley municipalities, the value of emission parameters can be adopted. As MSW open burning study is very primitive in Nepal, more research and development are indispensable. Moreover, site specific waste combustible fraction experiment is recommended for future research work.

Although Government of Nepal enforced SWM act 2011 for better SWM practices, it has not achieved satisfactory result yet. Thus, the result of this study is likely to become a stronger foundation for SWM policies that valley municipalities including other urban areas can implement in the immediate future. Amending the act and incorporating the most effective MSW open burning control strategy (e.g., increasing waste collection services, improved waste segregation at the source, improved waste collection systems, banning MSW open burning in the community level, high penalty charge for use-throw practices, extending more Kawadi centres for waste recycling, systematic waste disposal practices, air curtain incineration practices, and awareness) are the foremost steps to reduce this pollution and its effects.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.wasman.2018.08.013.

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Emission factors and emission inventory of diesel vehicles in Nepal

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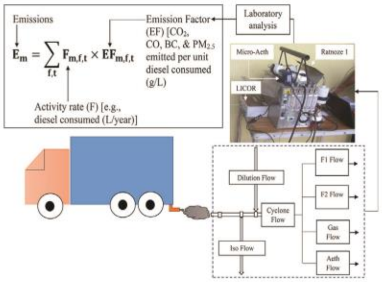
HIGHLIGHTS

- Experimentally-based EFs are used for emissions inventory of Nepal's diesel vehicles.
- In moving vehicles, EF of CO, BC, and $PM_{2.5}$ increased three-four times that of idle.
- The $PM_{2.5}$ emissions in 2017/18 were 6–7 times higher than in 1989/90.
- On average, $PM_{2.5}$ emissions from truck/tipper > bus > pickup > mini bus > micro bus.
- Revised transport policy with a low-carbon transport pathway can reduce emissions.

GRAPHICAL ABSTRACT



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ABSTRACT

A comprehensive emission inventory of the transport sector through fuel-based emission factors (EFs) was developed for the first time in Nepal. This study estimates air pollutants emission from diesel vehicles between the years 1989 and 2018 based on national statistical data, average vehicle kilometers travelled, fuel mileage, and measurement-based EFs for each vehicle category during idle and moving conditions. The consumption of diesel by vehicle category was also estimated and total consumption was compared with national sales data. The Monte Carlo was used to estimate uncertainties. Nationally, total diesel consumption was estimated as 892,770 kL (85–115%) in 2017/18, 13.4 times higher than 1989/90. Ratnoze1 and Microaeth were used to conduct the tail pipe emission measurements. The fuel-based EFs of CO_2, CO, BC, and $PM_{2.5}$ were calculated through the carbon mass balance method. Of all diesel vehicles measured ($n = 29$) during idling, the average EFs were estimated as CO_2 2600 (99–101%), CO 33.3 (44–156%), BC 0.6 (25–101%), and $PM_{2.5}$ 5.2 (0–235%) in unit of $g\,L^{-1}$. For moving conditions ($n = 5$), the average EFs were estimated to be CO_2 2476 (90–110%), CO 97.3 (0–232%), BC 1.7 (46–110%), and $PM_{2.5}$ 20.7 (0–255%), all in $g\,L^{-1}$. Multiplying fuel consumption by EFs, national air pollutant emissions were estimated as 2214 (90–110%) to 2781(85–115%) for $CO_{2,}$ 27.7 (42–158%) to 88.8 (0–232%) for CO, 0.51 (23–177%) to 3.55 (46–110%) for BC and 3.42 (0–236%) to 23.47 (0–255%) for $PM_{2.5}$ in 2017/18 in unit of Gg. This paper recommends revising national vehicle mass emission standards based on the findings of this study and including and enhancing sustainable low-carbon transport through amendment of transport policy.

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1. Introduction

The transport sector is one of the fastest growing and most recognized sources of deteriorating air quality (Guo et al., 2016). According to Singh et al. (2017), the number of motor vehicles has been doubled in every decade in Asia. Of total carbon dioxide emissions, 23% (Ribeiro et al., 2007) to 26% (Chapman, 2007) emissions are from the transport sector globally. The rise in air pollution from this sector is due to the rapid growth of motor vehicles, insufficient public transport and improper management, chaotic urban development, road congestion, low-grade engines, poor quality of fuel (Badami, 2005; Pucher et al., 2007), and lack of timely repair and maintenance (Mool et al., 2020). In the coming decades, a larger proportion of air pollution is likely to result from the transport sector (Franco et al., 2013). Transport sector emits greenhouse gases such as carbon dioxide (CO_2), nitrous oxides (N_2O), methane (CH_4) as well as harmful pollutants like carbon monoxide (CO), particulate matters ($PM_{2.5}$), black carbon (BC), hydrocarbon (HC), oxides of nitrogen (NO_x) and sulfur dioxide (SO_2) (Garga et al., 2006; Kathuria, 2004; Neeft et al., 1996; Raux, 2004; Ribeiro et al., 2007; Sadavarte et al., 2019; Singh et al., 2017; Wang et al., 2011; Wu et al., 2017). Though heavy-duty diesel vehicles (HDDV) proportion is lower, their emissions contribute significantly to air pollution problems (Grigoratos et al., 2019; Kirchstetter et al., 1999). According to Shrestha (2018), road vehicle emissions contribute in acid deposition, stratospheric ozone depletion and climate change. Severe air pollution can also results allergies, respiratory disease, cardiovascular disease, and premature deaths (Das et al., 2018a; Das et al., 2020).

The transport sector is increasingly being recognized as one of the important leading sources of air pollution for the Kathmandu Valley and Nepal. Despite the emerging recognition of vehicular emissions, it is very difficult to estimate emission inventory because of large uncertainties especially due to emission factors (EFs) (which largely depends on vehicle category, fuel quality, vehicle inspection and maintenance conditions) and other factors like total distance travelled, fuel consumed, mileage, driving conditions, climatic factors, load factors, age of the vehicles (Bellasio et al., 2007; Clark et al., 2002; D'Angiola et al., 2010; Jamriska et al., 2004; Jing et al., 2016; Mool et al., 2020; Wang et al., 2008), and road-grade (Cui et al., 2017). Although few literatures were reported for the Kathmandu Valley and Nepal, there were no studies that considered country-specific EFs for estimation of emission inventory from diesel vehicles. The study on on-road vehicle exhaust emissions was carried out in Kathmandu more than two decades ago, and the study was mainly focused on CO and HC emissions only (Zhang et al., 1995). Shrestha et al. (2013) performed a research on the examination of the vehicle fleet in the Kathmandu Valley with the purpose of estimating the environment and climatic co-benefits of technological incursions; nevertheless, the study was based on the International Vehicle Emission (IVE) model. Sadavarte et al. (2019) also published an emission inventory for the transportation sector; however, the analysis was solely based on secondary data, and the EFs were computed using global-based values.

For the first time in Nepal, comparative research on the creation of fuel-based EFs for idle and moving diesel vehicles was examined. The emissions inventory of gas phase pollutants (CO_2 and CO) and particulate matter (BC and $PM_{2.5}$) from stationary and moving vehicle categories (bus, mini bus/truck, truck/tipper, pickup, and micro bus) was studied using experimentally-based EFs, national statistical data, average vehicle kilometers travelled (VKT), and fuel mileage.

2. Material and methods

2.1. Sample size

The diesel vehicle emission measurement campaign started from 8 August 2019 until 30 January 2020. In total, thirty-four vehicles were studied, which included eleven buses, two mini buses, two trucks, five mini trucks, two tippers, one mini tipper, seven pickup, and four micro buses. Twenty-nine vehicles were measured while idling and five while moving. In idle condition, ten were buses, one was a mini bus, two trucks, three mini trucks, two tippers, one mini tipper, six pickup, and four micro buses. While in moving condition, one was a bus, one was a mini bus, one was a pickup, and two mini trucks.

2.2. Instrumentation

In the literature, a number of devices have been employed to measure pollutants (ARAI, 2007; Cui et al., 2017; McCormick et al., 2000; Pujadas et al., 2004; Robert and Kleeman, 2007; Singer and Harley, 1998; Singer and Harley, 2000; Zhang et al., 1995). Ratnoze 1 and Micro-Aeth AE51 were utilized to determine EFs in this investigation. Ratnoze 1 was used to determine EFs by the carbon balance method. It uses sensors for real time measurement of gaseous and particulate pollutants. The sample train contained a $PM_{2.5}$ size selective inlet and two parallel 47 mm filter holders that were used for collecting filter samples for laboratory analysis. Ratnoze 1 was fitted with several sensors to measure various pollutants such as CO (through electrochemical sensor), CO_2 (NDIR sensor), and PM (through light absorption, Micro-Aeth AE51) (Mountain Air Engineering, 2016). Likewise, Micro-Aeth AE51 measures real-time BC using a filter-based light absorption method. BC mass concentration during the operation was recorded in the non-volatile memory. The Micro-Aeth AE51 pump was adjusted to a sample flow rate of 50 mL/min, and data was logged every 10 s, which was then converted to a minute in accordance with Ratnoze 1 (AethLabs, 2016).

2.3. Sampling procedure and laboratory works

2.3.1. Preparation before a sampling event

Both the polytetrafluoroethylene (PTFE) and quartz filter paper were prepared for the sampling event. Quartz filters were dehydrated in a desiccator for 24 h after being maintained in a hot air oven at 310 °C for around 8 h. To eliminate moisture from PTFE filters, they were desiccated for 24 h. We have maintained the temperature and humidity condition of the filter paper before sampling. The initial weight of the filter papers was weighed using a digital balance, then placed in a sterile petri dish, labelled properly and sealed with Teflon tape.

2.3.2. Instrument calibration

The particulates and gaseous sensors of the Ratnoze 1 were factory calibrated. The device was verified using a calibration gas mixer before sampling. Prior to tailpipe emission measurements, the only onsite zero calibration was done utilizing a high efficiency particulate air (HEPA) filter. BC was measured using a factory-calibrated Micro-Aeth AE51. The CO gas sensors were tested at 0 ppm with a zero air gas (Specialty Gases Ltd., West Bromwich, UK) and at 81 ppm with a CO calibration gas mixture (Alchemic Gases and Chemical Pvt. Ltd., Mumbai, India). Similarly, a CO_2 calibration gas mixture was used to validate the CO_2 sensor at 350 ppm and 1250 ppm (Specialty Gases Ltd.). During pre and post field experiments, CO_2 data was collated using LICOR (LI-820, Lincoln, Nebraska, USA) for cross-checking reasons similar to the previous study (Adhikari et al., 2019).

2.3.3. Pre-sampling setup

Filter papers were loaded into their respective filter holders. A vacuum gauge was used to perform a leak test on the main inlet and dilution inlet prior to tail pipe emission measurement. To prevent background PM from collecting on the sample filters, the HEPA filter was attached to the cyclone intake. The serial data output was connected to Ratnoze 1 laptop using USB cable and Ratnoze 1 software was run. After loading a new filter in the MicoAeth AE51, the Ratnoze's valve was opened, and the equipment was turned on. Micro-Aeth AE51 was connected to its laptop and the software was run. To warm up the sensor box, it was turned on for 10 min. All the flow sensors (F1, F2, gas flow, dilution flow) were made zero and calibrated. The barometric pressure shown in the Kestrel meter was noted, which was then entered as the absolute stack pressure in the Ratnoze 1 calibration parameters. At the cyclone's tip, the Ratnoze1 pulled the exhaust sample mixed with dilution air at a flow rate of 1500 standard cubic centimeters per minute (sccm). At a flow rate of 400 sccm, the flow was then dispersed into channels F1 and F2. For the third channel, which led to the MicroAeth AE51, a flow rate of 50 sccm was maintained. Finally, the flow rate in the fourth channel, which was used to monitor gaseous and particulate pollutants, was controlled to maintain a total flow rate of 1500 sccm.

2.3.4. Exhaust sample period

The real-time concentrations of $PM_{2.5}$, BC, CO, and CO_2 along with background concentrations of CO and CO_2 were measured with the aforementioned instruments. Mool et al. (2020) discuss the significance of using background concentrations. The sample train consisted of $PM_{2.5}$ size selective inlet and two parallel 47 mm filter holders in which filter samples were collected for further laboratory analysis. Two filter papers viz. Pall Corporation Teflo filter membranes and quartz fiber filter membranes (of 47 mm diameter with 2 μm pore size), were used to collect the aerosol particles within the filter holders for gravimetric analysis.

In every tail pipe emissions measurement, a Kestrel meter was used to obtain air pressure value that was considered in the calculation of EFs. Depending on the mass of $PM_{2.5}$ deposited on the filter papers and field experience, the emissions from diesel vehicles were tested for more than 15 min for idle vehicles and about 5 min for moving vehicles. All sensor flows and the Micro-Aeth AE51 attenuation (ATN) were monitored and adjusted as needed during the sampling time. When the ATN value surpassed 120, the filter of Micro-Aeth AE51 was replaced. To keep all parameters within the detectable range, the dilution ratio (the ratio of dilution gas to sample flow) for all measurements was kept around 6 similar to Adhikari et al. (2019). To eliminate vapor impact on the measurement, the air used in dilution was dried in the silica chamber and blended with the exhaust sample.

2.3.5. Post-sampling

The filter papers were removed from the filter holders, put in the labelled petri dish and properly sealed with Teflon tape before being placed in the zip lock bag. The filters were then desiccated for 24 h. The temperature and humidity levels were maintained almost constant during the filter weighing process. Using a 0.01 mg resolution HPBG-2285Di micro-balance (BEL Engineering, Monza (MB), ITALIA), the concentrations were determined gravimetrically by subtracting the initial weights from the final weights.

2.4. Calculation

We looked at vehicles under 20 years old, taking into account the government of Nepal's vehicle prohibition policy from 2000 onwards (DoTM, 2012), as well as the life cycle of vehicles. From 1989/90 onward, data on vehicle registration was published. According to the five-year growth scenario, vehicles were on the road for at least six years before the government began publicly registering them. As a result, the vehicles ages were calculated starting in 1983/84. Each year's data was deducted from the original year after 20 years. In order to represent true diesel vehicles, electric trolley buses were also omitted from the data. The data on diesel vehicle registrations from 1989/90 to 2017/18, obtained from the government of Nepal's DoTM (1989–2018) annual reports, was then redistributed into Euro grade types, referring to an estimated proportion of Pre-euro (1989/90–1998/99), Euro I (1999/90–2011/12), and Euro III (> 2012/13). Euro II vehicles were not included in this analysis since there are no records of them. In the Fig. 1, the percentage of Euro-grade vehicles is indicated.

The average vehicle kilometers travelled per year (VKT/yr) was multiplied by the average fuel consumption (l/km) to calculate annual diesel consumption per vehicle category. An open interview with the vehicle driver as well as odometer readings were used to determine the value of the short and long route average VKT (Table S3). Bajracharya and Bhattrai (2016) and Bajracharya and Bajracharya (2013) provided the average mileage of micro buses and pickup. The average mileage of buses, mini buses, trucks, and mini trucks was not reported for Nepal, therefore it was considered to be from a neighboring country, India (Karali et al., 2019). The average mileage of a truck/mini truck was calculated using the truck and mini truck's reported values (Table S4). The carbon mass balance method was used to calculate the average fuel-based EFs of the vehicle categories. Mool et al. (2020) and Adhikari et al. (2019) describe the carbon mass balance method and comprehensive EF calculations using Ratnoze 1. The carbon mass balance approach was also used to determine fuel-based EFs for pollutants in another study in Nepal (Stockwell et al., 2016). In this method, the EFs are normalized to fuel consumption (i.e., grams of pollutants emitted per liter of fuel consumed) (Singer and Harley, 1996; Stockwell et al., 2016).

By carbon balance, it is possible to relate the amount of pollutants emitted to the amount of fuel burned if the molar exhaust concentrations of CO_2, CO, and BC are measured (Adhikari et al., 2019; Liu et al., 2009; Mool et al., 2020; Singer and Harley, 1996; Wang et al., 2011; Westerdahl et al., 2009). CO_2, CO, and BC are the primary carbonaceous products that are emitted from the combustion process. We did not consider OC or HC in this study. Neglecting it can have a minor effect on the calculation because gas-phase carbonaceous products are influenced by CO_2 (Ning et al., 2008; Wang et al., 2011; Westerdahl et al., 2009; Yli-Tuomi et al., 2005).

$$EF_P = \left(\frac{[P]}{\text{Total Carbon}}\right) * \frac{w_c \rho_f M_P}{12} \tag{1}$$

where, EF_P is the EF of pollutant P in unit of grams of pollutants per unit volume of fuel consumed, [P] is the exhaust concentration of pollutant P, w_c is the carbon weight fraction of the fuel or fuel carbon content, ρ_f is the fuel density and M_P is the molecular weight of P. The factor of 12 is the atomic mass of the carbon.

The EF of CO_2 and CO was calculated as follows:

$$EF_{CO_2}(g/L) = \left(\frac{\text{Exhaust concentration}(C_{CO_2})}{\text{Total Carbon}}\right) * \frac{w_c * \rho_f * 44}{12} * 1000 \tag{2}$$

$$EF_{CO}(g/L) = \left(\frac{\text{Exhaust concentration}(C_{CO})}{\text{Total Carbon}}\right) * \frac{w_c * \rho_f * 28}{12} * 1000 \tag{3}$$

where, $w_c = 87\%$ and $\rho_f = 0.832\ \text{kg L}^{-1}$. The w_c value was not reported for Nepal, therefore we considered the USA-based value (Kirchstetter et al., 1999), which was used in other studies of Nepal (Adhikari et al., 2019; Mool et al., 2020). Likewise, the average value of ρ_f for the BS III standard was considered from India (CPCB, 2010).

The EF of $PM_{2.5}$ and BC were calculated with reference to the EF of CO. EF_{CO} was converted to EF for fine particle mass (EF_{PM}) by the ratio of filtered PM mass and the corresponding mass of CO drawn through the filter (Jayarathne et al., 2018).

$$EF_{PM} = \frac{PM(\mu g/m^3)}{\text{Emitted concentration }(C_{CO})} * EF_{CO} \tag{4}$$

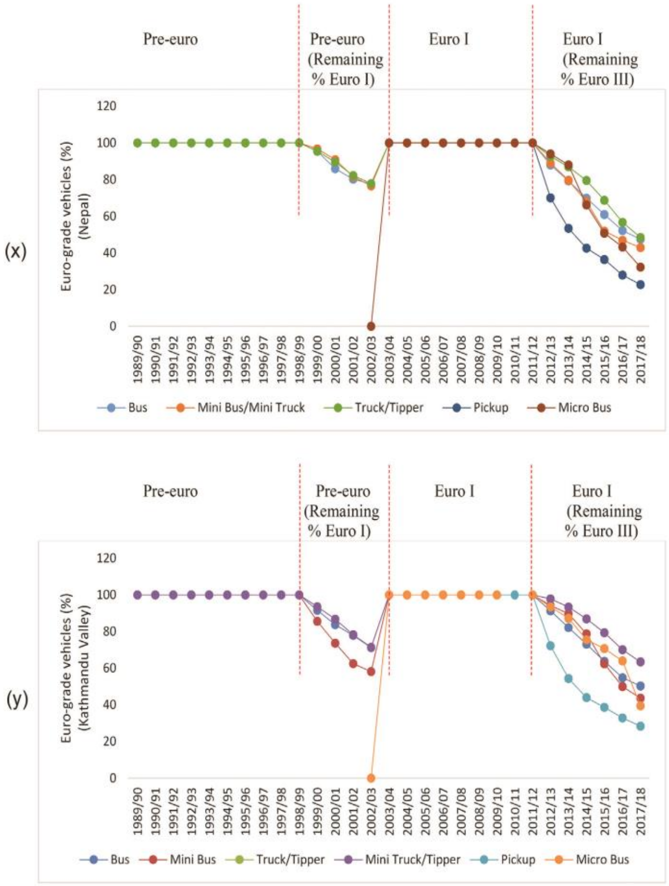

Fig. 1. (x-y). Re-distribution of Euro-grade vehicles. (Bagmati zone/Kathmandu Valley and Nepal).

$$EF_{BC} = \frac{BC(\mu g/m^3)}{\text{Emitted concentration } (C_{CO})} * EF_{CO} \tag{5}$$

Total carbon was calculated as follows:

$$\text{Total carbon } (\mu g/m^3) = C_{CO_2} + C_{CO} + BC \tag{6}$$

where, C_{CO_2} is the carbon concentration in CO_2 ($\mu g/m^3$); C_{CO} is the carbon concentration in CO ($\mu g/m^3$); and BC is the carbon concentration in BC ($\mu g/m^3$).

Since the concentration of CO_2 and CO was measured in ppm, it was converted into $\mu g/m^3$ using the ideal gas equation $PV = \left(\frac{m}{M}\right)RT$.

Where P denotes the pressure of the gas; V the volume of the gas; m the mass of the gas; M the mass of the gas per mole; R the universal gas constant (i.e., 8.314 J K^{-1} mol^{-1}); and T the temperature of the gas.

The exhaust concentration of carbon in CO_2 and CO is calculated as follows:

$$C_{CO_2}(\mu g/m^3) = \frac{CO_2(ppm) * \text{Pressure (Pa)} * 12}{R\left(JK^{-1}mol^{-1}\right) * \text{Temp (K)}} \tag{7}$$

$$C_{CO}(\mu g/m^3) = \frac{CO(ppm) * \text{Pressure (Pa)} * 12}{R\left(JK^{-1}mol^{-1}\right) * \text{Temp (K)}} \tag{8}$$

where, Temp is the Nozzle temperature.

With the help of the MS Excel tool, a series of data logged at one second time base from Ratnoze 1 was converted into 1 min on average. Likewise, it was applied for data obtained from Micro-aeth AE51.

It was impossible to monitor pollutants emission from trucks/tippers and micro buses while they were driving. As a result, it was calculated using similar vehicle categories as well as accessible literature. The lower reported value of CO_2 EF and the upper reported value of CO EF of truck/tipper were calculated using a mini truck/tipper, whereas BC EF and $PM_{2.5}$ EF (upper value) were calculated using the idle condition of the truck/tipper from the experiment (Nepal) and the idle and moving condition from Park et al. (2011), which was based in the United States. By referring to the proportionate values from the idle condition, the EF of $PM_{2.5}$ at moving conditions was calculated for pickup. Furthermore, it was impossible to undertake pollutant emission measurements from micro buses while they were driving. In this case, we considered the EFs of pollutants from mini buses (idling and moving circumstances) and micro buses (idle condition) to estimate EFs for moving conditions.

From the overall EFs, it was further categorized into Euro I (~ Bharat Standard, BS I) and Euro III (~ BS III) vehicles. Due to the lack of Pre-euro grade vehicles during the field study, EFs were calculated using the scaling factor (SF), which was applied to estimate annual trend of emissions. As shown in the table S9, the SF for each Euro grade vehicle was calculated using the reported regional and worldwide EFs. ARAI (2007) provided the EF of CO_2 and CO for buses, trucks, and light commercial vehicles (LCV) in the Pre-euro condition. The CO_2 EF for buses and LCV (Euro I) was obtained from the literature (ARAI, 2007; Gurjar et al., 2004), while the EF for trucks was obtained from ARAI (2007). CO_2 EF was taken from Yang et al. (2016) for buses (Euro III), whereas LCV was calculated proportionally from bus, and truck was assumed to be the same for buses. CO EF for Euro I bus was considered from ARAI (2007), Mashelkar et al. (2002), and Bajracharya and Bhattrai (2016). CO EF was calculated for Euro III buses using proportionate truck. The CO EF for trucks (Euro I) was obtained from ARAI (2007) and for Euro I and III trucks from Mahesh et al. (2019). Likewise, CO EF for LCV for Euro I was collected from ARAI (2007), Mashelkar et al. (2002), and Bajracharya and Bhattrai (2016). The CO EF for LCV (Euro III) was estimated by referring to proportionate data from trucks.

BC EFs for buses (Pre-euro) were determined proportionately from trucks (Kirchstetter et al., 1999). BC EF of diesel-LCV (Pre-euro) was calculated using the average value of buses and trucks. The published value of HDDV (Allen et al., 2001) was used to calculate the EF of BC for buses (Euro I) and Euro III (Raparthi and Phuleria, 2012). Geller et al. (2005) and Ban-Weiss et al. (2008) provided the BC EF for trucks (Euro I), whereas the BC EF for Euro III trucks and LCV was calculated using proportional data from buses (Raparthi and Phuleria, 2012; Allen et al., 2001). By reference to the average value of buses and trucks, the BC EF for LCV (Euro I) was calculated. ARAI (2007) and Pandey and Venkataraman (2014) provided the EF of $PM_{2.5}$ in buses and LCVs (Pre-euro and Euro I), whereas the EF of $PM_{2.5}$ in trucks was assumed to be the same as in buses. Moreover, a linear regression model was used to determine the $PM_{2.5}$ EF of Euro III vehicles.

The emissions from diesel vehicles were calculated from transport modes by referring to the equation as below (Pandey and Venkataraman, 2014).

$$E_m = \sum_{f,t} F_{m,f,t} \times EF_{m,f,t} \tag{9}$$

where, E_m is the total emissions in Gg yr^{-1} from mode m, and $F^{m,f,t}$ and $EF^{m,f,t}$ are the fuel consumption in MT yr^{-1} and EF in g kg^{-1} fuel burned respectively, for fuel type f and combustion technology t within that mode.

A total of 23,575 diesel vehicles were surveyed in seven different sites around the Kathmandu Valley to determine daily emissions (Fig. S1). The daily emissions were estimated using those vehicle category proportions.

2.5. Emission uncertainties

The Monte Carlo method was used to analyze the uncertainties of the findings. The coefficient of variation (CV) for activity data was calculated by referring to estimated diesel consumption and the average value of the minimum and maximum diesel consumption. The mean and standard deviation (SD) were calculated using the sets of available data similar to other studies (Das et al., 2020; Ni et al., 2015; Zhang et al., 2019; Zhao et al., 2011; Zhou et al., 2017). The range of fuel consumption, EFs, and emissions were calculated for idle and moving conditions by averaging 20,000 Monte Carlo simulations with a 95% confidence interval converted to a percentage of the mean (Table S14).

3. Results and discussion

3.1. Traffic condition

In Nepal, 3,221,042 vehicles have been registered between 1989/90 and 2017/18. In 2017/18, the total number of diesel vehicles (buses, mini buses/trucks, trucks/tippers, pickup, micro buses) was 270,723, rising 16.4 times from 1989/90 (DoTM, 1989-2018) (Table S1). From 1989/90 to 2017/18, pickup grew the most (96.3 folds), followed by micro buses (33 folds), trucks/tippers (12.7 folds), mini buses/trucks (12.4 folds), and buses (12.5 folds). From 1989/90 to 2017/18, 1,172,413 (or 36.4%) vehicles were registered in the Bagmati zone (DoTM, 1989-2018). By 2017/18, diesel vehicle registrations had risen to 75,436, a 22.5-fold increase from 1989/90 (DoTM, 1989-2018) (Table S2).

3.2. On-road vehicle flow

Mini buses made up the majority of the total number of vehicles observed, followed by pickup, micro buses, buses, trucks/tankers, mini trucks/tankers, tippers, construction vehicles, and mini tippers (Fig. S2). The mini bus (544 h^{-1}) had the greatest average vehicle flow per hour during the day (7 a.m. to 6 p.m.), followed by pickup (478 h^{-1}), micro bus (360 h^{-1}), bus (255 h^{-1}), truck/tanker (150 h^{-1}), mini truck/tanker (140 h^{-1}), tipper (25 h^{-1}), construction vehicles (9 h^{-1}), and mini tipper (4 h^{-1}).

3.3. Diesel consumption

Diesel consumption across Nepal has increased over the past three decades (Fig. 2; Table S5). Total national consumption was approximately 893,000 kL (85–115%, i.e., uncertainty ranging in between 759,000 kL and 1,030,000 kL) in 2017/18, 13.4 times the value in 1989/90. Of that amount, 55.9% was consumed in diesel vehicles (Table S5). After the transport sector, diesel consumption was highest in diesel generator sets (30–40%) for generating electricity, especially during the electricity crisis in Nepal (The World Bank, 2014). In 2008/09, the agricultural sector accounted for 19.87% and the industrial sector for 1.37% (WECS, 2010). In 2007/08, diesel fuel sales dropped by 1.3% despite a 11.7% increase in the number of diesel vehicles compared to the previous year. Similarly, in 2015/16, sales of petroleum products dropped significantly due to a severe earthquake and disruption in the fuel supply because of protests along the Nepal-India border (Underwood et al., 2020). Zooming in from the national scale to the Bagmati zone (a deprecated geopolitical region which contained all of Kathmandu Valley), we note that diesel consumption by vehicles increased by a factor of 17 between 1989/90 and 2017/18 with total estimated consumption at 262,000 kL in the final year (Fig. 2; Table S6).

The estimated diesel usage was compared to the government of Nepal's publicly available reports (NOC, 1993-2018). This study's value was 10.5% lower in 2008/09 than WECS (2010), but 12.5% higher in 2011/12 than WECS (2014) (Fig. 2). It was also compared to the most recent literature (Sadavarte et al., 2019). For 2001/02 (> 6.6%), 2005/06 (> 4.2%), and 2016/17 (3%), the anticipated amount was near. This study, on the other hand, found that the value of 2011/12 was 25% and 12.5% higher than Sadavarte et al. (2019) and WECS (2014), respectively. It should be noted that this study excludes other diesel-powered vehicles like as construction and ambulances/jeeps/cars, which could have influenced the results slightly.

3.4. Emission factors

In this study, fuel-based EFs were employed since they had less variance in terms of driving mode, vehicle weight, and engine power than travel-based EFs (Fu et al., 2012; Kean et al., 2003; Mool et al., 2020; Park et al., 2011). The country-specific EFs of pollutants from diesel vehicle categories were calculated in this study (Fig. 3; Table 1; Fig. S3–4). The average EFs of the entire diesel vehicles measured (n = 29) during idling condition were CO_2 2600 (99–101%), CO 33.3 (44–156%), BC 0.6 (25–101%), and $PM_{2.5}$ 5.2 (0–235%) in unit of g L^{-1}. In the same way, average EFs for moving situations (n = 5) were calculated as

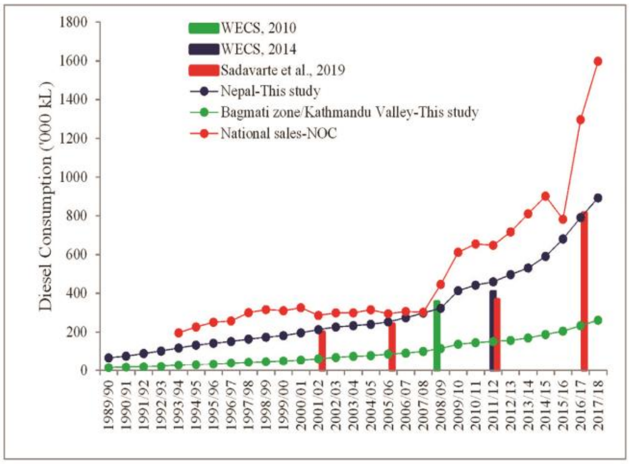

Fig. 2. Yearly trends in total diesel consumption. (Bagmati zone/Kathmandu Valley and Nepal).

CO_2 2476 (90–110%), CO 97.3 (0–232%), BC 1.7 (46–110%), and $PM_{2.5}$ 20.7 (0–255%), all in g L^{-1}. The EFs were estimated as CO_2 2559–2605, CO 30.01–58.8, BC 1.03-1.59 and $PM_{2.5}$ 5.92–15.31 for bus, mini bus (CO_2 2271–2574, CO 50–237, BC 0.6-2.2, $PM_{2.5}$ 1.6–5.4), mini truck/tipper (CO_2 2554–2591, CO 39.6–62.5, BC 0.7-1.9, $PM_{2.5}$ 5.5–54.5), truck/tipper (CO_2 2564–2602, CO 32.9–96.5, BC 0.4-2, $PM_{2.5}$

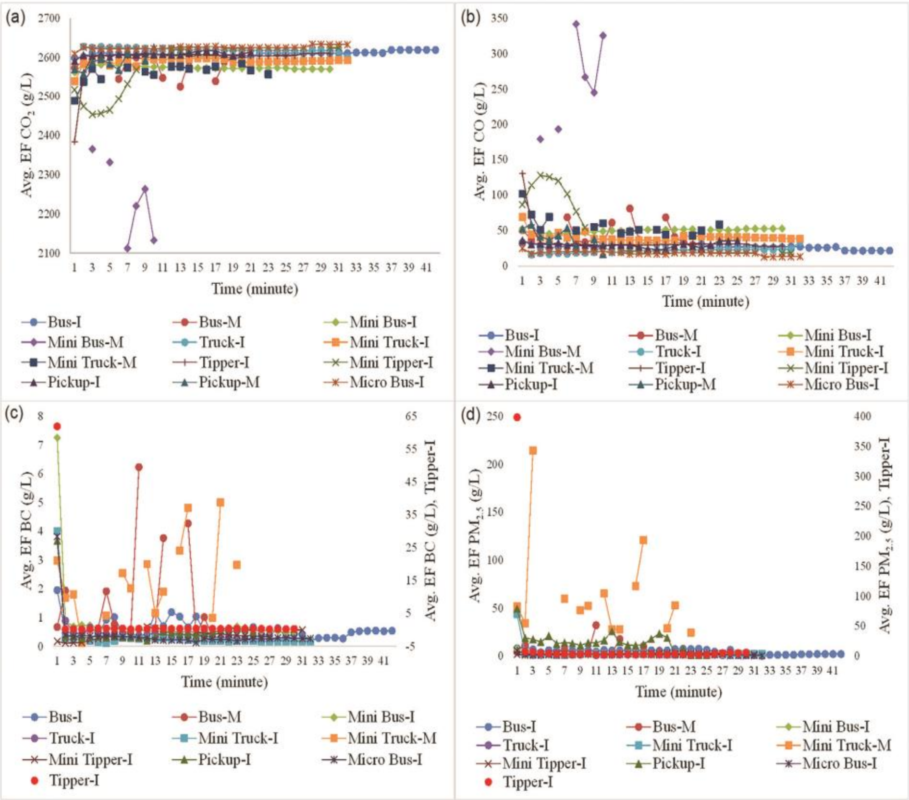

Fig. 3. (a-d). Emission Factors by vehicle category. Note: I represents vehicles in idling condition and M represents vehicles in moving condition.

Table 1
Nepal and global emission factors (g kg^{-1}).

References	Year	Vehicle category	CO_2	CO	BC	$PM_{2.5}$
This study	2019	Truck/Tipper-I, Nepal	3025	38.2	0.5	4.1
This study	2019	Truck/Tipper-M, Nepal	2982	113	2.7	19.2
Mool et al., 2020	2017–2018	Tipper-I, Nepal	3137	33.0	0.4	6.9
Cui et al., 2017	–	HDDT-I, China	–	–	–	0.5
Wei et al., 2017	2014	HDDT-I, China	–	–	0.2	–
Park et al., 2011	2007	HDDT-I, USA	–	75.0	0.2	0.4
Mool et al., 2020	2017–2018	Truck-I, Nepal	3101	56.0	1.3	37.1
Liu et al., 2009	2007–2008	LDDT-M, China	–	–	–	0.6
Wang et al., 2011	2009	HDDT-M, China	–	42.0	2.2	2.35
Huo et al., 2012	2007 and 2011	HDDT-M, China	–	29.0	–	1.4
ARAI, 2007; Sadavarte et al., 2019	–	Truck, India	3291	–	–	–
Subramanian et al., 2009	2008	LDDT & HDDT-D, Thailand	–	–	–	8.4
Ban-weiss et al., 2008	2008	HDDT-M, USA	–	–	0.9	1.4
Burgard et al., 2006	2005	HDDT-M, USA	–	32.0	–	–
Bishop et al., 2001	1997–1999	HDDT-M, USA	–	31.0	–	–
Schneider et al., 2008	2005	HDDT-M, Germany	3190	–	0.2	–
This study	2019	Pickup-I, Nepal	3034	32.4	0.5	14.8
Mool et al., 2020	2017–2018	Pickup-I, Nepal	3149	24.0	1.8	17.6
This study	2019	Pickup-M, Nepal	3019	41.6	1.0	29.2
This study	2019	Bus-I, Nepal	3029	34.9	1.2	6.9
Mool et al., 2020	2017–2018	Bus-I, Nepal	3107	52.0	0.8	11.9
This study	2019	Bus-M, Nepal	2976	68.4	1.8	17.8
ARAI, 2007; Sadavarte et al., 2019	–	Bus, India	3467	–	–	–
This study	2019	Micro BusI, Nepal	3050	22.8	0.4	2.4
This study	2019	Micro Bus-M, Nepal	2690	108.3	0.4	8.5
Kirchstetter et al., 1999	1997	HDDV-M, USA	–	–	1.3	2.5
Weingartner et al., 1997	1993	HDDV-M, Switzerland	–	–	0.3	–
This study	2019	Mini BusI, Nepal	2994	58.2	0.7	1.8
This study	2019	Mini Bus-M, Nepal	2641	275.7	2.6	6.3
This study	2019	Mini Truck/Tipper-I, Nepal	3013	46	0.8	6.4
This study	2019	Mini Truck/Tipper-M, Nepal	2970	72.6	2.2	63.4
This study	2019	Mini Bus/Truck-I, Nepal	3050	22.8	0.4	2.4
This study	2019	Mini Bus/Truck-M Nepal	2690	108.3	1.5	8.5

Note: I = Idling condition, M = Moving condition.

3.5–16.3), pickup (CO_2 2596–2610, CO 27.8–35.8, BC 0.4-0.9, $PM_{2.5}$ 12.7–25.2), and micro bus (CO_2 2314–2623, CO 19.7–93.1, BC 0.4-1.3, $PM_{2.5}$ 2.1–7.3), all in g L^{-1}. In the same way, the table S10 and S11 shows the Euro grade-wise EFs based on the measurement and estimation from gathered literature.

The EF of pollutants per vehicle category calculated in this study differs from those reported in previous studies in Nepal, India, China, the United States, and Germany (Table 1). Of the four pollutants, high variations in EF were noted for BC and $PM_{2.5}$. We were able to gain a basic understanding of the EFs by comparing data from various vehicle categories within the country and around the world. The findings of this study showed all vehicles did not have an influence of age on EFs. Furthermore, EFs varied considerably across automobiles of the same generation. As a result, robust generalizations could not be established by merely connecting vehicle age and EFs; additional factors must be taken into account. Those factors include vehicle categories/types and age-distributed vehicle population (Park et al., 2011), road-way grade (Cui et al., 2017), fuel composition/quality (Cui et al., 2017; Wang et al., 2011; Wei et al., 2017), poor driving condition (i.e., low and high speed) (Cui et al., 2017; Wang et al., 2011), design of engine, operating conditions, manufacturing company, vehicle maintenance pattern, traffic conditions, and environmental conditions like temperature, humidity, pressure, and altitude (Mool et al., 2020).

The majority of diesel-powered vehicles in Kathmandu are old and poorly maintained (Ale and Nagarkoti, 2003; Mool et al., 2020). Low vehicle speeds (< 20 km h^{-1}, observed value) and old vehicles with high mileage, coupled with narrow and hilly roads in the valley. Pierson et al. (1996) conducted a prior tunnel research that looked at the influence of highway slope and engine load on exhaust emissions. The study found that driving uphill on a 3.8% grade roughly doubled CO and NOx EFs expressed per unit distance travelled and increased the EF of volatile organic compound (VOC) by 50% as compared to downhill driving on a 0.6–3.8% grade in the Fort McHenry Tunnel. More recently, emissions in a Swedish tunnel with both uphill and downhill parts were shown to be dependent on driving circumstances, with EFs in grams per kilometer increasing by a factor of up to ten during crowded traffic times compared to smooth driving conditions (Sjodin et al., 1998).

3.5. Emissions trend

On average CO_2, CO, BC, and $PM_{2.5}$ emissions from diesel vehicles have all increased in recent years. In the 2017/18 fiscal year, national air pollutant emission were estimated to be 2214 (90–110%) to 2781 (85–115%) for CO_2, 27.7 (42–158%) to 88.8 (0–232%) for CO, 0.51 (23–177%) to 3.55 (46–110%) for BC and 3.42 (0–236%) to 23.47 (0–255%) for $PM_{2.5}$ in 2017/18 in unit of Gg. CO_2 was 13.1–16.1 folds, CO (5.1–5.3) folds, BC (3.0–4.9) folds, and $PM_{2.5}$ (6.0–7.5) folds respectively higher than in 1989/90 (Figs. 4-5; Table S12). There was a sharp decline in the emissions trend of CO, BC, and $PM_{2.5}$ in the years 2003/04. One of the possibilities is that before 2003/04, old vehicles (pre-euro grade) dominated the vehicle fraction (90–100%) along with higher EFs. From 2003/04 onwards, entirely Euro I were introduced, having a lower EF than the Pre-euro grade (Fig. 1). In contrast, CO_2 EF did not change significantly, even though vehicles were getting older, so the emission trend did not fluctuate. The emissions from diesel vehicles in 2017/18 were compared to total emissions (including diesel and gasoline) from the transportation sector (Sadavarte et al., 2019). Our findings show that CO_2 emissions were fairly similar; however, CO emissions were 1.7–6.7 times lower, while BC and $PM_{2.5}$ emissions ranged in between 0.4 and 4.5 and 0.2–1.5 folds respectively compared to that of the literature (Table 2). CO_2 emissions in the Bagmati zone/Kathmandu Valley were estimated to be 638–680 Gg, CO (9.7–26.2 Gg), BC (0.2–0.8 Gg), and $PM_{2.5}$ (1.0–6.5 Gg) in 2017/18, higher by 16.7–17.1,

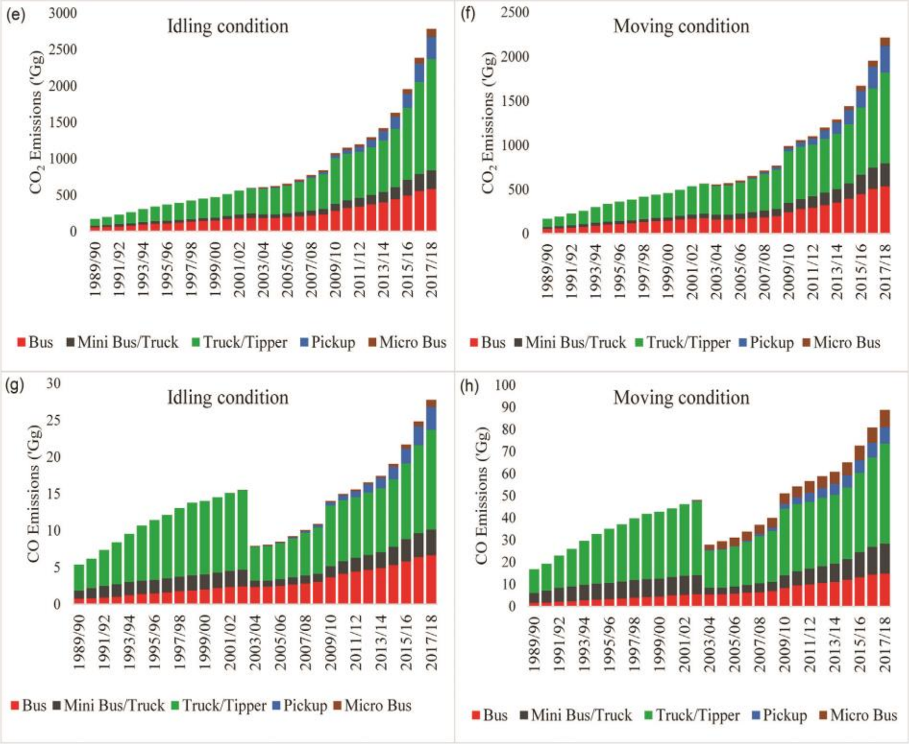

Fig. 4. (e-h). Trends in CO$_2$ and CO emissions in Nepal by vehicle category.

5.1–6.6, 4.4–4.9, and 6.6–8.9 times, respectively, than in 1989/90 (Table S13). From the available literature, the emissions of diesel vehicles were compared to overall emissions (i.e., diesel and gasoline) (Table 2). The calculated CO$_2$ value was 1.4–1.5 and 2.0–2.2 times lower in contrast to Shrestha et al. (2013) and Ghimire and Shrestha (2014), respectively. CO emissions increased by 0.6–2.0 and 0.4–1.2 times, respectively, as compared to Shrestha et al. (2013) and Ghimire and Shrestha (2014). The BC and PM$_{2.5}$ values were 0.7–7.0 and 0.3–2.6 times lower than Shrestha et al. (2013), respectively. Trucks/tippers contributed the most emissions in Nepal in 2017/18, while micro buses contributed the least (Fig. 6, m-t).

The emissions by vehicle category for the Bagmati zone/ Kathmandu Valley are highlighted in the Fig. S5 (e-h). During 2017/ 18, daily CO$_2$ (138.3–147.2 tons) and PM$_{2.5}$ (241.3–1449.2 kg) were at their peak around 2–3 p.m., whereas BC (36.8–178.3 kg) and CO (2.1–5.8 tons) were at its highest around 11 a.m.-12 p.m. (Fig. 7; Fig. S6 (i-l)).

The difference in the emissions could be due to variations of selecting parameters like vehicle categories, age-distributed vehicle population, survival fraction of vehicles, VKT, and fuel consumption. According to previous studies, high traffic congestion, poor fuel quality, and old & inadequate road vehicle maintenance all led to a large increase in transportation-related emissions (Ale and Nagarkoti, 2003; Das et al., 2018b; Mool et al., 2020). Carbon deposits form in the cylinders and injectors when a vehicle is driven at a low speed (< 2000 rpm) amid congested traffic. Even if the vehicle is new, this degrades engine performance and results in excessive emissions (Faiz et al., 2006). This study's fuel quality test revealed that 80% of samples exceeded the sulfur level of BS III (350 mg L^{-1} as per Nepal Oil Corporation). Only two samples met the BS II requirements (compatible with EURO II). Furthermore, lead (Pb) was found in diesel fuel (Table S15). All of this data points to low fuel quality in Nepal. Furthermore, other published studies used EFs that were totally secondary in nature and not from Nepal, which might have influenced the results.

3.6. Air quality management opportunity

It is critical to look into cost-effective solutions that might cut emissions considerably. The initial measures in reducing air pollution are to repair and maintain roads, improve fuel quality (> BS III standard), and purchase high-quality (> IV euro grade) vehicles. Another research (Mool et al., 2020) found that timely service of diesel vehicles can lower emissions by 1.4 times BC and 2.9 times PM$_{2.5}$. National Climate Change Policy (2019), National Environmental Policy (2019), and Second Nationally Determined Contribution (NDC, 2020) envisage for the promotion of high grade, electric, hybrid, hydrogen-powered, and other clean fuelled vehicles. The concept of climate-resilient economic growth via a reliable transportation sector is currently a top focus. The Department of Transport Management's vehicle emission testing program must be expanded to all of Nepal's provinces and implemented effectively. The Nepal air quality management action plan (NAQMAP) is critical in recognizing the serious issue about air pollution. Subsidies, tax exemptions, and the scrapping of high-emitting vehicles are all necessary actions for promotion and meeting the intervention objective, but the country is falling short. According to Das et al. (2018b), highly polluting obsolete transport vehicles should be phased out gradually through scrappage or incentive schemes.

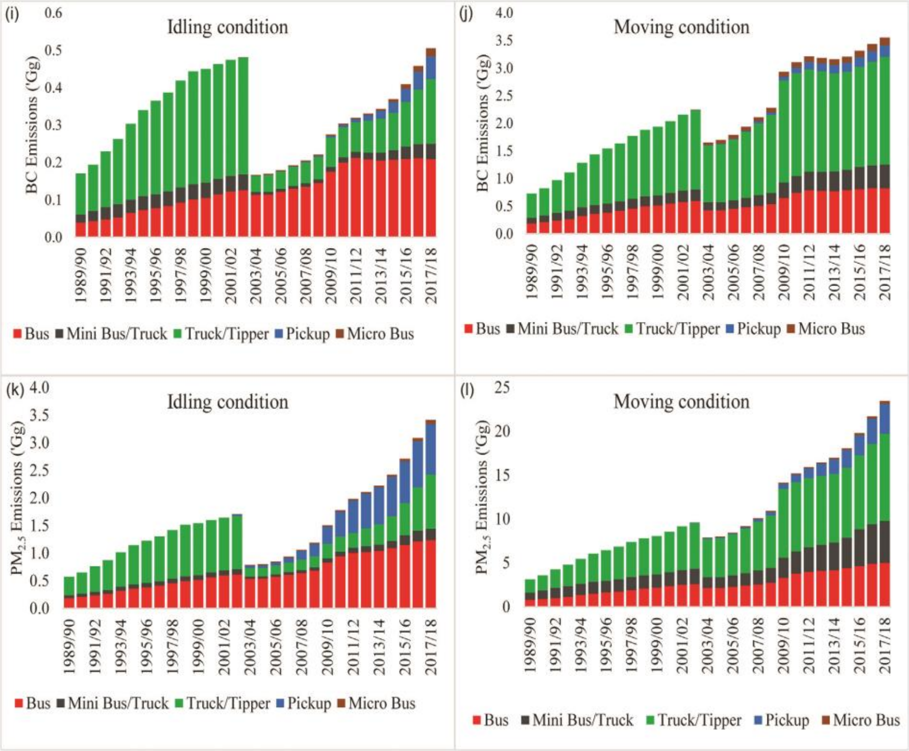

Fig. 5. (i-l). Trends in BC and $PM_{2.5}$ emissions in Nepal by vehicle category.

3.7. Study limitation

This study used 29 idling samples that might well reflect the EFs of various diesel vehicle categories. However, due to the small sample size, temporal EFs could not be calculated. Only 5 samples were evaluated for moving vehicles to understand the range of EFs and emissions in relation to idling circumstances. The ratio of EF of BC and $PM_{2.5}$ in 50% of the samples was between 20% and 95%, while the remaining 50% of the samples indicated <20%. The Micro-Aeth AE51 equipment used in this work for BC concentration measurement was verified with another high-tech BC measuring device (Aethalometer), with R^2 showing 98%. As a result, it's plausible to assume that there are a number of controlling elements that influence the EF of BC. Those variables were not identified or quantified in this investigation. Despite the studies performed over the previous three decades, one controlling issue, according to Salako et al. (2012), is the lack of a universally acknowledged standard technique to quantify BC.

All of the findings in this study are based on instruments that are now accessible (Ratnoze 1 and Micro-Aeth AE51) and carbon balance methods that have been published in prior publications. Further study with bigger sample numbers is needed to get a reliable generalization of uncertainties conclusion.

Table 2
Comparative estimates of emissions (in Gg).

	Year	CO_2	CO	BC	$PM_{2.5}$
Nepal					
This study	2008/09	770–840	10.9–39.9	0.2–2.3	1.2–10.9
Sadavarte et al., 2019	2008	782	69.37	0.89	1.8
Kathmandu Valley					
This study	2010/11	1057-1150	15.0–54.2	0.3–3.1	1.8–15.2
Shrestha et al., 2013	2010	1554	31	2.117	4.685
This study	2014/15	1441-1628	19.1–65.1	0.4–3.2	2.4–18.1
Ghimire and Shrestha, 2014	2014	3196	23.81	–	–

4. Conclusions

This study used experimentally-based EFs measured in Nepal to build a comprehensive diesel vehicle emission inventory for the country from 1989/90 to 2017/18. The EFs were computed using the carbon balance method for five vehicle categories: bus, mini bus, truck/tipper, pickup, and micro bus. In the idling condition, the average EFs were estimated as CO_2 2600 (99–101%), CO 33.3 (44–156%), BC 0.6 (25–101%), and $PM_{2.5}$ 5.2 (0–235%) in unit of $g\ L^{-1}$, whereas in the moving condition, CO_2 2476 (90–110%), CO 97.3 (0–232%), BC 1.7 (46–110%), and $PM_{2.5}$ 20.7 (0–255%), all in $g\ L^{-1}$. Total emissions of CO_2, CO, BC, and $PM_{2.5}$ grew substantially from 1989/90 to 2017/18. Nationally, air pollutant emission were estimated as 2214 (90–110%) to 2781(85–115%) for CO_2, 27.7 (42–158%) to 88.8 (0–232%) for CO, 0.51 (23–177%) to 3.55 (46–110%) for BC and 3.42 (0–236%) to 23.47 (0–255%) for $PM_{2.5}$ in the year 2017/18 in unit of Gg. Vehicle categories, age-distributed vehicle population, survival proportion of vehicles, VKT, fuel consumption/mileage, engine efficiency, and EFs are some of the key elements that impact variance in emissions. During the morning peak hour (9–10 a.m.), CO emissions in the Kathmandu Valley were highest due to mini buses, which comprised 9% of all vehicles and had the highest EF (50 g CO/L while idling and 237 g CO/L while moving). More EF studies with a larger sample size are required

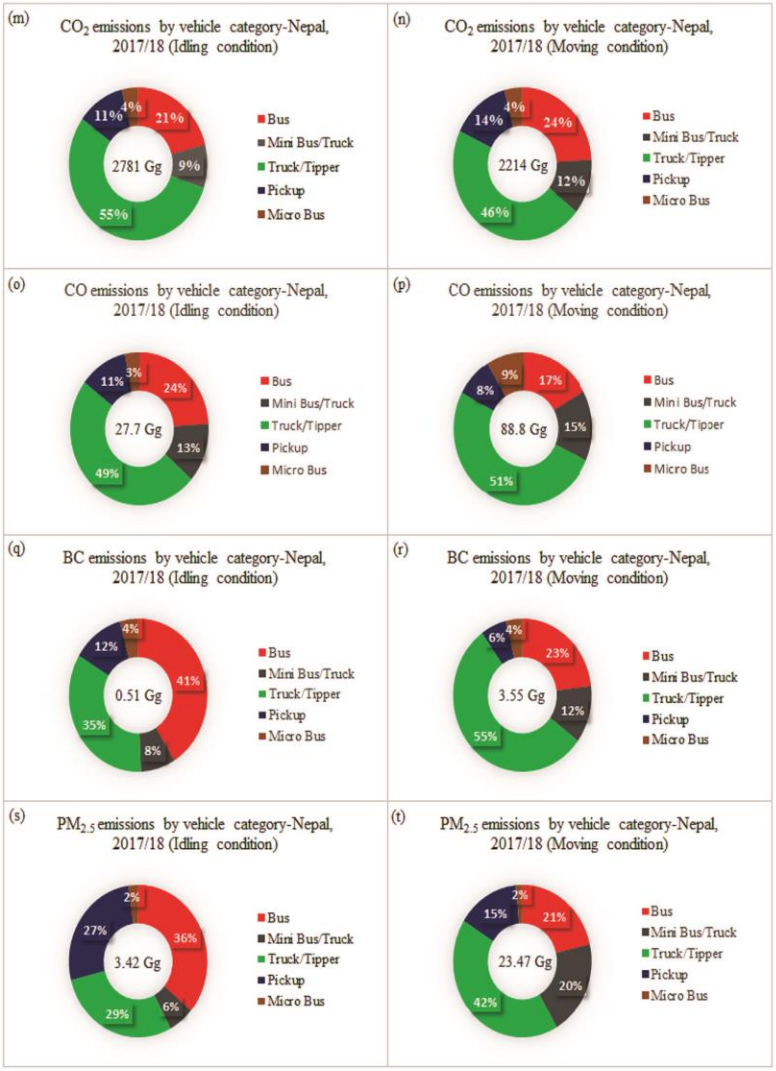

Fig. 6. (m-t). Emissions by vehicle category in Nepal.

to reduce uncertainty in our inventory estimates. Only the Kathmandu Valley's daily emissions were examined in this study. Other regions of the country are likely to have different fleet characteristics, for which further research is necessary. Based on the findings of this study, it is recommended that national vehicle mass emission standards (i.e., CO_2, CO, BC, and $PM_{2.5}$) be revised and effectively implemented. It also suggests amending the 2058 B.S. transportation policy to incorporate and promote sustainable, low-carbon transportation for better air quality management.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.152539.

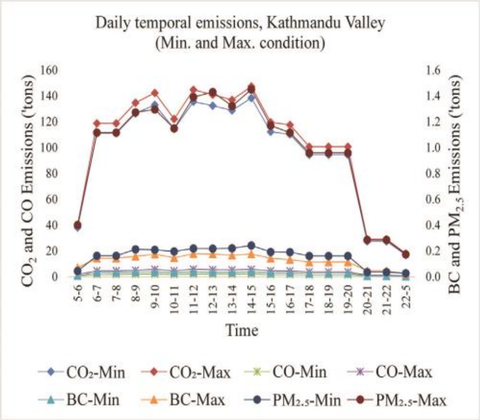

Fig. 7. Daily temporal emission-Bagmati zone/Kathmandu Valley.

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Traffic Condition and Emission Factor from Diesel Vehicles within the Kathmandu Valley

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ABSTRACT

Past research on air quality within the Kathmandu Valley indicates that diesel vehicles make a substantial contribution to the ambient pollution. Hence, it's important to identify cost-effective measures for reducing their emissions. As a first step, roadside observations of diesel vehicles were recorded between February and April 2017 at six locations: two on the ring road (RR), two inside the RR, and two on major arterial highways outside the RR. Out of all diesel vehicles observed (n = 12,039), 35% were emitting a visible plume of black smoke and hereafter are referred to as "superemitters". Of the 4,248 superemitters, 45% were buses of varying sizes, 34% were large trucks, and 19% were small pickups. Superemitters made up the largest fraction of diesel vehicle traffic on the RR (43%–46%) but were also abundant inside the RR (27%–29%), where human population and pollutant exposure is greatest. Upon developing a comprehensive understanding of the superemitting vehicle types and ownership, maintenance patterns and servicing costs were studied through a survey of vehicle owners, vehicle drivers, and local maintenance centers. The costs of general servicing ranged between USD 16 for tractors and USD 203 for construction vehicles depending on the size of the vehicle. Lastly, the effect of general servicing on emissions while idling was explored for a small sample of superemitters (n = 4). $PM_{2.5}$ emissions reduced from 10.90 g L^{-1} to 3.76 g L^{-1} and BC emissions reduced from 0.847 g L^{-1} to 0.596 g L^{-1} after servicing. Taken together, results from this roadside surveillance study and exploratory emission-measurement campaign provide preliminary evidence that a policy of mandatory, routine maintenance of a targeted subset of the diesel fleet can systematically reduce emissions and improve air quality in the Kathmandu Valley and other cities around the world that are facing similar problems.

Keywords: General servicing; Plume opacity; Tailpipe exhaust; Urban transportation; Roadside surveillance.

INTRODUCTION

Air pollution levels are extremely high in urban areas of low- and middle-income countries (LMIC), where they represent an enormous burden on public health. Extensive research has been conducted in the urban areas of LMIC in recent decades (Edgerton *et al.*, 1999; Molina and Molina, 2004; Petkova *et al.*, 2013; Wang *et al.*, 2013; Sahu and Kota, 2017), but the pace of air quality improvement remains slow in many of those cities (Colbeck *et al.*, 2010; Maji *et al.*, 2015; Njoku *et al.*, 2016). Among those cities, the Kathmandu Valley is one such highly polluted area in the South Asian region (Parajuly, 2016; Mahapatra *et al.*, 2019). Earlier, air quality studies were limited in scope, especially because of the absence of appropriate instruments for measurements in the valley. This had caused inaccurate quantification of the total pollution load. However, in recent years, high-quality measurements of air quality have been started to address this issue.

A recent study conducted by Mahapatra *et al.* (2019) indicates an increasing trend of aerosol loading within the Kathmandu Valley by approximately 50–60% in between 2000–2015. This high pollution load could be attributed to rapid population and vehicular growth along with the increase in the energy demands of the valley. At the same time, the diurnal and seasonal variations in the pollutants of the valley were strongly influenced by the changes in the vehicle fleet (Sharma *et al.*, 2012). Meanwhile, few source apportionment studies pointed out that emissions from vehicles, brick kilns, residential combustion, waste/biomass burning and soil dust, were the major contributors to pollution in the valley (Stone *et al.*, 2010, 2012; Kim *et al.*, 2015; Sarkar *et al.*, 2017; Shakya *et al.*, 2017). All these

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recent studies suggest that vehicular emissions is one of the major contributing factors to the ambient air pollution throughout the year (Sarkar *et al.*, 2017; Shakya *et al.*, 2017). On top of this, vehicular traffic is growing most rapidly, and in the period 2006–2016, it rose at a rate of 17.94% (Department of Transport Management, 2017). In 2004–2005, particulate matter with an aerodynamic diameter of less than 10 microns (PM_{10}) from diesel vehicles contributed to about 27% of the valley's emissions (Gautam, 2006). This fraction increased to about 34% of the total PM_{10} emissions in 2015–2016 (Department of Environment, 2017).

Not all the diesel vehicles plying the roads of the Kathmandu Valley contributes for pollution; in fact, a small proportion of high-emission vehicles (about 25% of the total diesel vehicle fleet) contributed to half of the emissions (Ale and Nagarkoti, 2003; Zhang *et al.*, 1995). The high-emission vehicles are responsible for increasing the deviation in the emission distribution, even though average emissions have decreased substantially (Bishop and Stedman, 2008). These high-emission vehicles can be referred to as "superemitters". Some previous studies have classified superemitter vehicles based on visual observations (McCormick *et al.*, 2003), while others have differentiated them based on probability distributions (Subramanian *et al.*, 2009).

To date, no study has specifically examined the composition and emission contribution of the superemitter vehicles among the entire diesel vehicle fleet in the Kathmandu Valley. Only limited studies on the characteristics of these vehicle fleets have been carried out. In 2014, a study was conducted through roadside observations and the manual count method in order to identify the different types of vehicles in the Kathmandu Valley (Ghimire and Shrestha, 2014). The study indicated that trucks, minitrucks and tankers predominated among the diesel vehicles (Ghimire and Shrestha, 2014). However, the study did not address the composition and fraction of the superemitter vehicles among the total diesel vehicle fleet in the valley. In this study, therefore, we have identified – in terms of vehicle type and vehicle owner – the contribution of the superemitter vehicles to pollution.

Given the significance of diesel vehicle emissions and their contribution to Kathmandu's air quality, it is important to understand the emission factors (EF) of individual vehicle types. To date, only one study has been conducted, and that too only on the EF of pollutants and the impact of general servicing of gasoline vehicles in the Kathmandu Valley during engine idling (Jayarathne *et al.*, 2018). The study indicated that the EF of $PM_{2.5}$ reduced by 92% after servicing (Jayarathne *et al.*, 2018); however, the values for diesel vehicles remain unknown. The emissions ($PM_{2.5}$ and black carbon, BC) from diesel vehicles were higher than those from gasoline ones (Kirchstetter *et al.*, 1999; Ban-Weiss *et al.*, 2008), and so we initiated a pilot study to understand the EF of pollutants from diesel vehicles. The $PM_{2.5}$, BC, CO and CO_2 EFs of different types of diesel vehicles were measured during idling, and we also tried to understand the influence of general servicing on the reduction of emissions from diesel vehicles.

Many initiatives have already sought to control emissions from diesel vehicles. One of these, the inspection of diesel vehicles, was ineffective (Ale and Nagarkoti, 2003; Faiz *et al.*, 2006). There are still not enough efficient technologies and human resources to carry out efficient vehicle testing in the valley (Jha, 2001; Gurung, 2016). Recently, the head of the Department of Transport Management (DoTM) publicly acknowledged that the inspection and maintenance program in Kathmandu is non-functional (Gurung, 2016). However, it's clear that the controlling of emissions from vehicles, including superemitters, can be addressed by cost-effective (Bond and Sun, 2005; Bhandarkar, 2013). The emissions from diesel vehicles after servicing were low compared to the vehicles that had not been serviced (Larssen *et al.*, 1997), and in the Kathmandu Valley, they reduced by 34–42% after servicing (Ale and Nagarkoti, 2003).

The present study, through roadside observations in the Kathmandu Valley, tries to understand the dominance of superemitter diesel vehicles with respect to vehicle types and vehicle owners. To our knowledge, it is the first study in Nepal and Kathmandu Valley that records, through measurements, the EFs from diesel vehicles and explores the impact of proper servicing on emissions. Earlier, numerous studies had been carried out to estimate the emission load of various pollutants from different vehicle types. These emission inventories were prepared using decades-old EFs from different countries (Shrestha and Malla, 1996; Shrestha *et al.*, 2013; Ghimire and Shrestha, 2014; Bajracharya and Bhattarai, 2016). This created a huge uncertainty (in terms of overestimation or underestimation) in the emission inventory of the Kathmandu Valley (Sarkar *et al.*, 2017; Mahapatra *et al.*, 2019). A realistic and updated emission inventory of the vehicle fleet in the valley has yet to be developed and there is a strong need to develop it by incorporating local estimates to reflect the real scenario of the area (Mues *et al.*, 2018; Mahapatra *et al.*, 2019).

This study provides the initial inputs into preparing an emission inventory of these vehicles; this will also help quantify global emissions. Further, it will provide evidence so that informed decisions can be taken in controlling the ambient air pollution in the Kathmandu Valley. An important point is that the EF calculated for diesel vehicles in this study, and the impact of servicing, do not represent all the diesel vehicles in the valley. In this regard, larger, in-depth studies have been conducted (Zhang *et al.*, 1995), along with regular monitoring. However, our study provides a baseline value for future research calculation of EF of diesel vehicles in the valley during engine idling, along with data on the impact of servicing on the EFs of these vehicles. In the future, we propose to carry out a detailed study of the emissions from different types of vehicles during idling, and when moving uphill and downhill, which will represent all diesel vehicles and driving conditions prevalent in the valley.

METHODS

Roadside Survey

A prominent feature of the Kathmandu Valley roadway network is a 27-km-long ring road (RR); it encompasses the urban centers of Kathmandu and Lalitpur, and carries two-way traffic, with four lanes in each direction. Numerous arterial roads emanate outward from the RR, while the two main arteries carry traffic to and from Bhaktapur (Eastern Highway) and the Trishuli River Valley (Western Highway). A web of local and residential streets, and a few multi-lane collectors, lie within the RR (Fig. 1).

Our survey locations were selected to characterize the wide variety of diesel traffic found in and around Kathmandu. The sites included two on the RR (Ekantakuna and Chabahil), two inside the RR (Kuleshwor and Sorakhutte), and one on each of the main arterial highways outside the RR (Ghathaghar and Kalanki). The exact locations met three selection criteria: (i) free-flowing traffic; (ii) minimal road

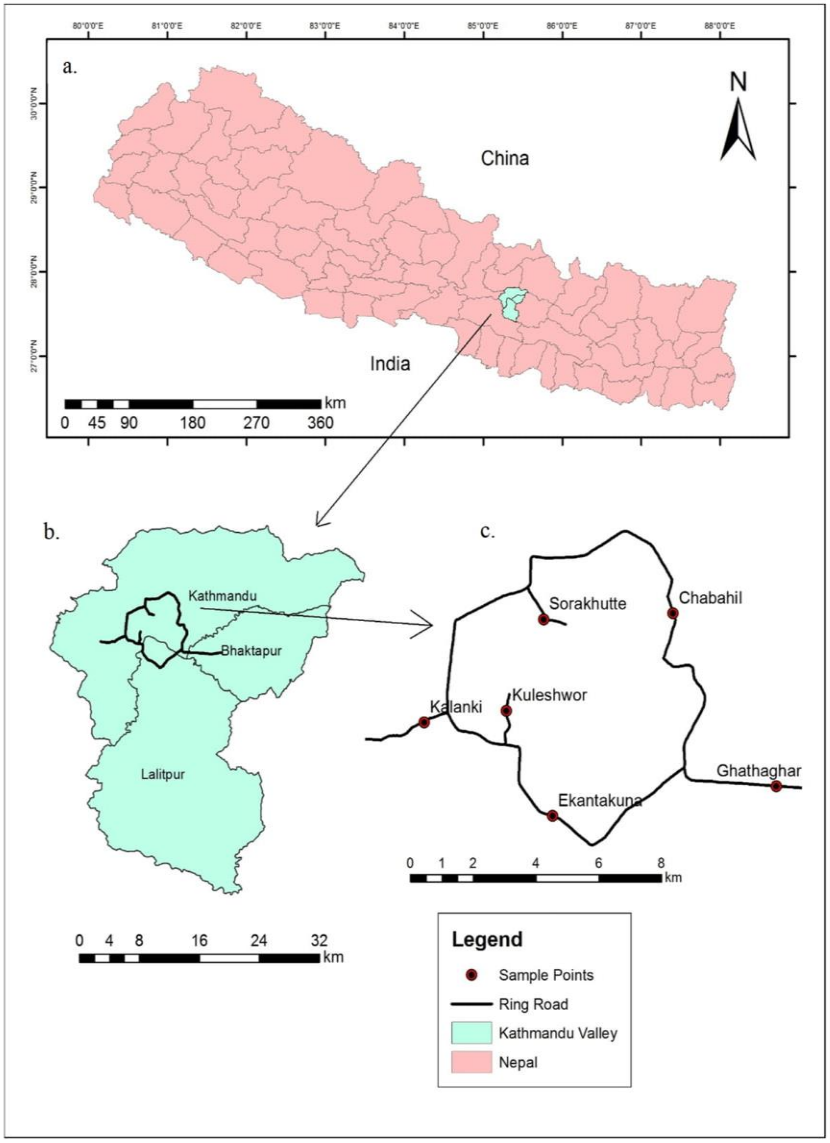

Fig. 1. (a) Map of Nepal indicating the study area (Kathmandu Valley); (b) Map of study area showing the ring road encircling Kathmandu and Lalitpur; (c) Sample points on the ring road, inside the ring road, and the arterial highways.

dust in the vicinity so that dust plumes would not obscure the surveyors' view of exhaust plumes; and (iii) either slight uphill gradients or the presence of speed breakers.

Among the six selected locations, three had a speed breaker on the road and the other three were on gently sloping stretches. The drivers had to either accelerate after crossing the speed breaker or increase their power output to go uphill. In both cases, the surveyors were afforded the opportunity to see whether the vehicle was emitting a visible plume of smoke.

Observations were collected on one working day at each site between February and April 2017 (Table 1). The data were recorded manually during daylight hours. Only diesel-powered vehicles were counted and only the traffic flow in one direction (i.e., uphill or immediately after the speed breaker) was considered. Superemitter vehicles were differentiated from normal vehicles if the plume of black smoke was visible behind the tailpipe of those vehicles.

Classification of Vehicles and Color of License Plates

Eight vehicle types were chosen to segregate the fleet by vehicle size and transport purpose (i.e., cargo versus passenger). The cargo vehicle types were truck (carrying goods other than construction materials), tipper (carrying construction materials like sand and gravel), tanker (carrying water and oil), pickup (light-duty truck having a gross vehicle weight of < 4 metric ton), tractor (that transports goods and also used on farms), and construction vehicle (e.g., roller and bulldozer). Because the counts were very low, construction vehicles and tractors were grouped together as a single vehicle type. Passenger vehicle types were bus (> 26 seats), minibus (16–25 seats), and microbus (< 15 seats) – all inclusive of the driver's seat.

In Nepal, the license-plate color distinguishes the type of owner each vehicle is registered to (Government of Nepal, 1993). In order to find some correspondence between vehicle owner and emission characteristics, the surveyed vehicles were further subdivided by the color of their license plates:

- Vehicles providing shared transport (e.g., buses and for-hire trucks) – black plate with white characters and white plate with black characters
- Privately owned vehicles – red plate with white characters
- Tourism vehicles (e.g., owned by hotels and tour operators) – green plate with white characters
- Government-owned vehicles – white plate with red characters
- Vehicles owned by national corporations (e.g., Nepal Electricity Authority, Nepal Telecom) – yellow plate with blue characters
- Diplomat-owned vehicles (e.g., United Nations fleet, Embassy employees) – blue plate with white characters

Maintenance Survey

Several surveys were conducted – between 17 April and 12 May 2017 – to characterize the patterns of diesel vehicle maintenance in the Kathmandu Valley. A total of 193 owners and drivers of different types of diesel vehicles were surveyed at Balkhu, Balaju, Kalanki, Kalimati, Ratnapark,

Table 1. Roadside survey locations with composition of superemitter vehicles.

S.N.	Locations	GPS coordinates		Direction	Slope (%)	Number of observed traffic lanes	Date	Local time		Diesel vehicle count	Superemitter vehicle (%)
		Latitude	Longitude					Start	Stop		
1.	Eastern Highway (Ghathaghar)	27.67367°N	85.37587°E	South West	1.7	2	19 March 2017	7:00	17:30	3176	28
2.	Western Highway (Kalanki)	27.69083°N	85.27496°E	South West [a]	6.6	1	16 March 2017	7:00	17:30	2513	40
3.	On RR1 (Ekantakuna)	27.66574°N	85.31175°E	South East [a]	2.7	1	27 February 2017	6:30	17:30	1919	43
4.	On RR2 (Chabahil)	27.72009°N	85.34626°E	South East [a]	5.1	1	17 March 2017	7:00	13:30	1762	46
5.	Inside RR2 (Sorakhutte)	27.71850°N	85.30922°E	North East	7.1	2	27 April 2017	7:00	17:30	1581	27
6.	Inside RR1 (Kuleshwor)	27.69398°N	85.29850°E	South	1.6	2	13 March 2017	6:45	17:30	1088	29
	Total									**12,039**	

S.N. represents serial number.

[a] represents presence of speed breaker to the direction of flowing vehicles.

RR represents ring road.

and Gongabu. In addition, 29 local maintenance centers were surveyed to get their perspectives on how closely Kathmandu drivers and vehicle owners adhered to the recommended maintenance schedules. Our initial conversations with the owners and drivers indicated a preference towards locally owned, unaffiliated workshops, and so we conducted surveys at 26 such workshops and with only 3 authorized dealers.

Measurement of EF and Calculations

The Ratnoze1 (Mountain Air Engineering, USA) portable sampling system was deployed at the maintenance centers from 28 August through 7 September 2017 to measure emissions from normal and superemitter in-use diesel vehicles under idling conditions. This system measures in situ concentrations of $PM_{2.5}$ (based on the scattering coefficient), CO, CO_2, and BC (Mountain Air Engineering, 2016). The system also measures the background concentrations of gaseous pollutants like CO and CO_2 in the dilution train. Generally, the EFs are derived from the concentration of CO and CO_2. The error would be less when the emission concentrations of these gaseous pollutants are higher than the background concentration. To address this issue, background corrections are important and need to be included. This is the standard procedure whereby the background corrections of the samples are made; this can also be found in a previous study conducted by Stockwell *et al.* (2016). Pall Corporation Teflo filter membranes and quartz fiber filter membranes (of 47 mm diameter with 2 µm pore size) were used to collect the aerosol particles within the filter holders. The mass scattering efficiency (MSE) was determined to compute the real-time concentration of $PM_{2.5}$. MSE is the ratio of $PM_{2.5}$ scattering coefficient to its mass concentration (Hand and Malm, 2007; Latimer and Martin, 2019). The scattering coefficient was obtained from the instrument, and the mass concentration from the filter samples by gravimetry for a given sampling period. Detailed descriptions of the equipment, flow movement and the calibration of sensors can be found in a previous study (Adhikari *et al.*, 2019).

The exhaust was sampled for 35–80 minutes depending on vehicle availability. For some vehicles, it was feasible to collect measurements before and after servicing. The servicing of vehicles included changing engine oil, and replacing oil, diesel and air filters. The main focus with respect to superemitter vehicles was on the data obtained from roadside observations. We made 14 observations and sampled 10 vehicles (Table 2).

The average and individual fuel-based EF were calculated from the data collected during the measurement system by using the carbon mass balance method. This method is based on the fuel combustion process in which the carbon mass emitted by the vehicle exhaust equals the carbon mass of the fuel consumed (Kirchstetter *et al.*, 1999; Moosmuller *et al.*, 2003). In this process, CO_2, CO and BC were considered to be the primary carbonaceous products emitted during the combustion process. The EFs were calculated using the equations reported in Adhikari *et al.* (2019).

RESULTS AND DISCUSSIONS

Composition of Diesel Fleet by Vehicle Type

A total of 12,039 diesel-powered vehicles were observed during our roadside survey, representing roughly a quarter of the on-road diesel vehicles registered in the Kathmandu Valley (Department of Transport Management, 2017). Our survey results (Fig. 2(a)) show that the on-road diesel fleet was relatively evenly divided between cargo carriers (25% pickup, 16% large truck, 9% tipper, 2% tanker, and 1% tractor and construction vehicle) and passenger buses (26% full-sized bus, 12% minibus, and 9% microbus). This observed vehicle mix was quite consistent with the vehicle registration data from the DoTM (Table S1). A slightly higher frequency of passenger vehicles (bus, minibus, and microbus) was noted in our survey than in the registration data, while the opposite was true in the case of cargo vehicles (e.g., pickup, tipper). A likely explanation is that the buses, on an average, travel a greater distance within

Table 2. Emission factors (g L^{-1}) of diesel vehicles tested during idling.

S.N.	Vehicle ID	Servicing status	Vehicle types	Model year (age of engine)	Emission factor (g L^{-1})			
					$PM_{2.5}$	BC	CO	CO_2
1.	V1	B	Truck	2002	30.80	1.067	46.2	2580
2.	V2	A	Truck	2008	05.82[a]	0.396[a]	22.1[a]	2619[a]
3.	V3	B	Bus	1998	15.50	1.358	81.3	2524
4.	V3	A	Bus	1998	06.64	0.899	81.4	2525
5.	V4	B	Bus	2012	07.29	0.460	20.4	2621
6.	V5	B	Bus	2013	10.10	0.949	28.6	2608
7.	V6	B	Bus	2016	07.88	0.253	46.4	2581
8.	V7	B	Bus	2016	08.56	0.248	40.8	2590
9.	V7	A	Bus	2016	03.03	0.211	48.6	2577
10.	V8	B	Tipper	2016	04.97	0.285	37.8	2594
11.	V8	A	Tipper	2016	02.67	0.282	31.9	2604
12.	V9	B	Tipper	2009	06.52	0.404	17.3	2626
13.	V10	B	Pickup	1999	14.60	1.494	20.3	2620
14.	V10	A	Pickup	1999	02.71	0.993	22.2	2618

Note: B, EF before vehicle servicing; A, EF after vehicle servicing (changing engine oil, and oil, diesel and air filters).
[a] represents only changing diesel filter; unable to obtain data before servicing.

Kathmandu than do cargo vehicles (Bajracharya and Bhattarai, 2016). Supporting this explanation at the other extreme were tractors and construction vehicles (e.g., excavators, cranes, and road-construction equipment) which are known to contribute very little to the fleet total of Vehicle Kilometres Travelled (VKT). These vehicles comprised only 1% of the diesel traffic at our survey sites, but constituted 6% of the registered vehicles (Table S1).

Considerable spatial heterogeneity in the mix of diesel vehicles can be seen in Fig. 2(b). At both the survey sites inside the RR (Sorakhutte and Kuleshwor), pickup trucks made up 38% and 42%, respectively, of the diesel traffic. This was noticeably higher than at the other four locations (17–25% pickups) where heavy-duty trucks and tippers were allowed to transport goods. The highest fraction of trucks was found along the Western Highway (30%) which is the primary route for cargo transport to/from India and southern Nepal. Trucks of this type were also quite common on the Eastern Highway (18%) and along the RR in Ekantakuna (21%). The highest frequency of tippers was

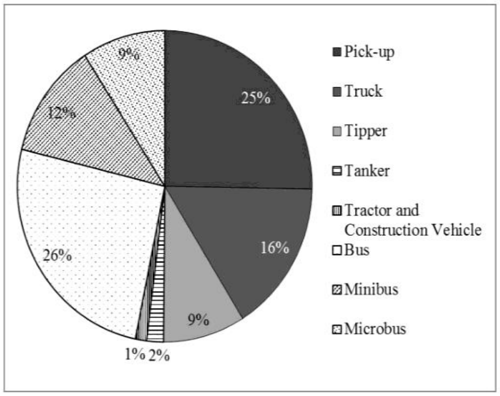

Fig. 2(a). Composition of all vehicles from roadside observations. The first five vehicle types represent cargo diesel vehicles; the remaining three types represent passenger diesel vehicles.

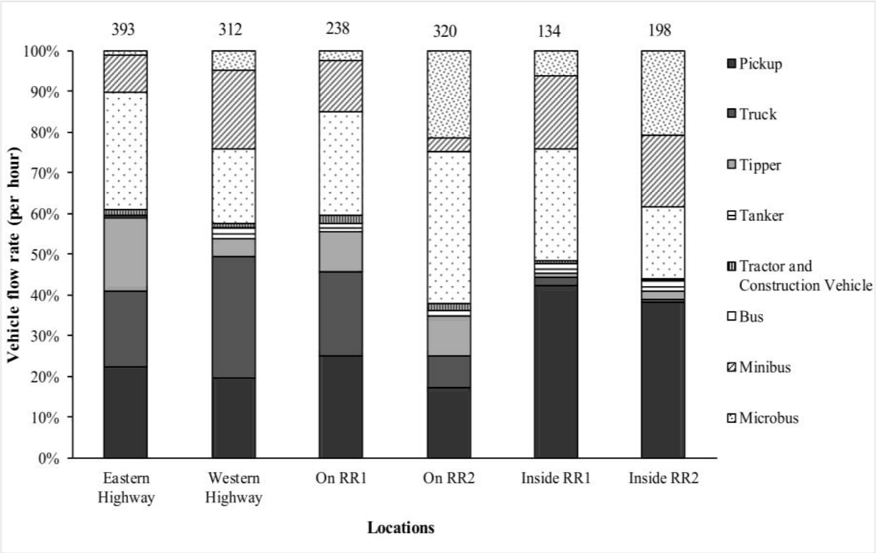

Fig. 2(b). Vehicle flow rate (h^{-1}) at each survey location listed above each bar for all diesel vehicles. Each data set is further subdivided into eight vehicle types (indicated by shading). The numbers above each bar represent the total number of vehicles observed at the particular location.

found on the Eastern Highway (18%) where the transport of construction materials is prevalent and a high concentration of brick kilns is located.

The highest fractions of buses and microbuses (37% and 21%, respectively) were along the RR in Chabahil owing to the close proximity of a bus depot. Passenger buses, as a whole, were also more than the number of cargo vehicles at both sites inside the RR, but not at the other three survey sites. Among the types of passenger buses classified in our survey, full-sized buses were the most common at all sites except on the Western Highway (where minibus presence was slightly greater at 19%) and inside the RR at Sorakhutte where microbuses were the most common (21%) owing to the vast network of narrow roads in proximity to that site.

The diesel vehicle flow rate was the highest on the Eastern Highway (393 h^{-1}), along the eastern segment of the RR in Chabahil (320 h^{-1}), and on the Western Highway (312 h^{-1}). The flow rate was moderate along the south-western segment of the RR in Ekantakuna (238 h^{-1}). The flow rate was the lowest inside the RR (198 h^{-1} in Sorakhutte and 134 h^{-1} in Kuleshwor), reflecting restrictions on heavy-duty truck traffic in the city center. One limitation of our study was that all the surveying was conducted during daylight hours so that the vehicles emitting visible plumes of soot could be distinguished accurately. Nevertheless, some informative temporal patterns in the diesel traffic were noticeable in our data set. At all the six sites, the flow rate of cargo vehicles peaked in the middle of the day (between 10:00 and 13:00) and was almost twice as high than in the morning hours (ca 07:00). However, bus traffic exhibited a completely opposite temporal profile, with flow rates at all sites reaching their minima during midday (between 11:00 and 13:30) (Figs. S2(a) and S2(b)). Peak flow rates were observed during either morning or evening commute hours – depending on the direction surveyed relative to the dominant flow of traffic – and were 2.2 times higher than the midday minimum.

Composition of Diesel Fleet by Vehicle Owner

Shared-transport vehicles constituted almost 65% of Kathmandu's diesel fleet (7,786 of 12,039) at our survey locations (Fig. 3(b)). As one might expect, most of these were buses of varying sizes. However, a large number of them were trucks (1,744) and tippers (946). Most of these vehicles were hired or leased for fares. Privately owned vehicles accounted for 31% of the diesel fleet and were predominantly pickups. The remaining license-plate colors were relatively uncommon in our survey: 1.6% government owned, 1.4% tourism related, 1.2% national corporation owned, and 0.12% diplomat owned (Table S2).

An attempt was made to corroborate the ownership information gathered from our roadside survey against the government data on vehicle registrations. However, we were unable to isolate the latter by fuel type (i.e., in terms of diesel-powered vehicles). For example, 3,750 blue-plate vehicles (i.e., owned by diplomats) were registered with the government; but most of them were fueled by petrol and that fraction had not been tabulated in Nepal's registration database.

Given the sizeable body of evidence summarized above, we concluded that the 12,039 diesel vehicles observed in our survey were characteristic of Kathmandu's vehicle fleet during February–April 2017. Therefore, it is appropriate to use this database to draw conclusions about the superemitting subpopulation of vehicles within Kathmandu's diesel fleet.

Prevalence of Superemitters in Kathmandu

Altogether, 4,248 diesel vehicles (i.e., 35% of the Kathmandu fleet) were visually identified as superemitters based on the plume of black smoke seen spewing from their tailpipes. Although this is a qualitative measure of the emissions, it lays a foundation for more quantitative studies in the future. Previous reports estimated that 20% of the diesel trucks and buses in Kathmandu were "smoke belchers" (Larssen *et al.*, 1997), but we could not find any systematic surveys for comparison with the present results.

An analysis of Kathmandu's vehicle inspection data (Ale and Nagarkoti, 2003) found that between 25% and 40% of the diesel fleet was responsible for half of the smoke emissions from April 2000 to April 2003, which is consistent with our roadside survey findings. More recent data from Kathmandu's vehicle inspection program are unreliable owing to the inadequate calibration of equipment at the test facilities (Gurung, 2016).

It is noteworthy that the superemitter fraction is much higher in Kathmandu (i.e., 35%) than in other countries. For example, an extensive review of relevant literature has it that about 20% of the diesel fleet in Asia and Latin America are superemitters (Bond *et al.*, 2004). In China, about 20% of the diesel fleet were responsible for half of the BC emissions in 2009 (Wang *et al.*, 2011). In the USA, this figure drops to less than 5% (Park *et al.*, 2011).

Influence of Speed Breakers

Table 1 suggests some spatial variability in the superemitter fraction: that is, it is most prevalent on the RR (43–46%) and less prevalent inside the RR (27–29%). This variability cannot be explained by spatial differences in the vehicle mix because it persists even when the same comparison is made for individual vehicle types. For example, the superemitter fractions for each of the eight vehicle types at the RR sites are higher than the corresponding fractions inside the RR.

Ultimately, the site-to-site variability in the superemitter fraction may be attributed to differing roadway characteristics. The fraction was higher at all sites next to speed breakers than at any of the sites where traffic was free-flowing (Table 1). This result held true for individual vehicle types as well. There is a statistically significant difference between the proportion of emitters in speed breaker and free-flowing road conditions. The implication is that the diesel vehicles in Kathmandu are more likely to belch smoke during acceleration (i.e., immediately after passing over a speed breaker) than when driving up a slope (1.6–7.1%) (Table 1).

Superemitters by Vehicle Type

It is evident that the vehicle types with the highest superemitting fractions were truck (51%), tanker (50%), and tractor and construction vehicle (48%); the three vehicle types with the lowest superemitting fractions were pickup (27%), minibus (27%), and tipper (29%) (Fig. 3(a)). Most of the trucks and tankers were old (Department of Transport Management, 2017) and did not comply with European Union norms. This has led to higher emissions from these types of old vehicles. However, tippers being the newly introduced heavy vehicles to the Kathmandu Valley fleet (Department of Transport Management, 2017) are compliant with Euro III norms, and have more advanced engine modifications than the older vehicles. Consequently, these vehicles were found to emit less. The majority of the tractor category included in this study had a small engine capacity, but high emissions. Usually, most of these tractors were old and had two-stroke engines. These types of engines are less efficient in fuel utilization and lack efficient lubrication systems, thereby exacerbating engine wear and tear, and increasing the emissions (Manufacturers of Emission Controls Association, 2014; Alves *et al.*, 2015).

When focusing on data from the sites with speed breakers, a significant difference was found between the microbuses at Chabahil (on RR2) and those at Kalanki (on the Western Highway). There was a higher number of superemitter microbuses at Chabahil. This might have been because, among the shared microbuses that are only used on the RRs, the majority were superemitters. The other influencing factors for this might also be traffic congestion and road conditions, which triggered a higher number of full-size superemitter buses at Chabahil compared to Kalanki.

At sites with free-flowing traffic, the most significant difference in emission characteristics was found between the full-size buses at Kuleshwor (in RR1) and those at Ghathaghar (on the Eastern Highway). Most of the full-size Nepal Yatayat buses at Kuleshwor were superemitters; but these were found to be absent at Ghathaghar. This might have resulted in a higher number of superemitter vehicles in Kuleshwor.

Superemitters by Vehicle Owner

The highest superemitter vehicles were shared-transport ones (39%) (Fig. 3(b)). This result was similar to those reported in the study by Ale and Nagarkoti (2003) in 2000–2001 and 2001–2002. One explanation may be that shared-transport vehicles are most extensively utilized, often to the point of overloading (Faiz *et al.*, 2006). In addition, people pay less to use these facilities, and so the owners are less concerned about maintaining their vehicles. The superemitting fraction was the lowest for tourism vehicles (16%) and diplomatic vehicles (0%). Tourism is one of the income-based sectors that provides and attracts its guests with the best facilities, and so tourist vehicles are clean and well maintained. The diplomatic vehicles are the highest standard vehicle type in Kathmandu's fleet and are maintained regularly. These two sectors do not favor old, inferior vehicles and prefer those with lower emissions.

For a few vehicle types, our data set is large enough to explore whether the emission characteristics are significantly affected by vehicle ownership. For example, we found that almost all national corporation tankers were superemitters (15 of 16), whereas the opposite was true of private tankers (5 of 31). Among the more common vehicle types, we should note that microbuses with black plates were twice as likely to be identified as superemitters (40%) than microbuses that were not shared-transport (20%).

Vehicle Servicing

Vehicle servicing was based both on travel distance and working period. Vehicle servicing (except for tractor and construction vehicle) was based on the distance travelled. The total cost of general servicing for each vehicle type included the cost of engine oil; oil, diesel and air filters; and labor. The average total cost of general servicing ranged between USD 16 for tractor and USD 203 for construction

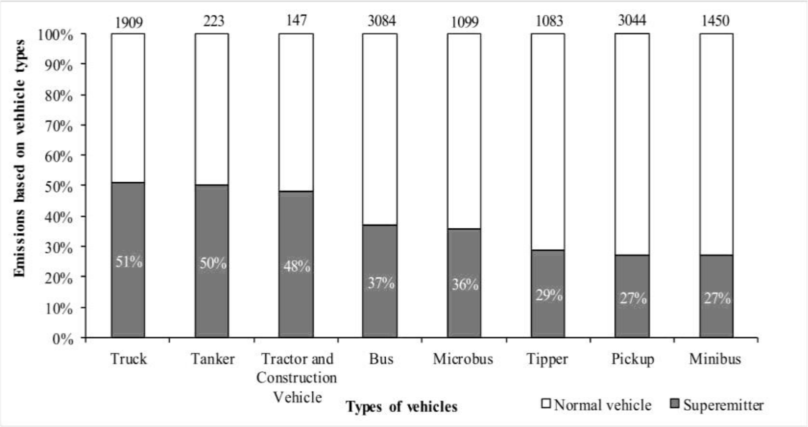

Fig. 3(a). Composition of superemitter vehicles based on vehicle types. The numbers above each bar represent the total number of individual vehicle types that were observed.

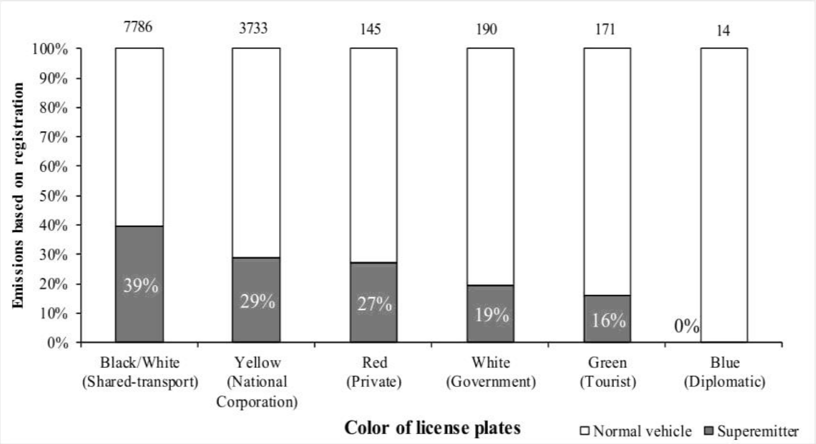

Fig. 3(b). Composition of superemitter vehicles based on registration. "Shared-transport" represents both the vehicles having black plates and white figures, and those with white plates and black figures. The numbers above each bar represent the total number of observed vehicles with particular color of license plates.

vehicle depending on the size of vehicle included in this study (Table S3). The cost of servicing varied according to the engine size of the vehicle. For example, the maximum total cost was for a construction vehicle with the highest engine capacity, and the minimum was for a tractor with the lowest engine capacity. The total costs of servicing were similar for all types of vehicle, according to the respondents from vehicle owners and local maintenance centers, with the difference ranging between 1% and 25%. In addition, most (71%) respondents were found to perform routine servicing of vehicles, of which 70% were normal (non-emitters). However, among 29% of the non-routinely serviced vehicles, 53% were superemitters.

EF of Diesel Vehicles

Fuel-based EFs were used in this study because their variation is comparatively less in terms of driving mode, vehicle weight and engine power compared to travel-based EFs (Kean *et al.*, 2003; Park *et al.*, 2011; Fu *et al.*, 2012). The fuel-based EFs of pollutants that were measured during idling are shown in Table 2. The EFs of CO, $PM_{2.5}$ and BC ranged between 17.3 g L^{-1} and 81.3 g L^{-1}, 4.97 g L^{-1} and 30.80 g L^{-1}, and 0.248 g L^{-1} and 1.494 g L^{-1}, respectively, before servicing (Table 2). It is notable that during idling of heavy diesel vehicles, the EFs of $PM_{2.5}$ and BC were higher than in other countries (Table 3). For example, the average EFs of $PM_{2.5}$ and BC for heavy diesel vehicles in this study were 18.62 g kg^{-1} and 0.827 g kg^{-1}, respectively. In a 2007 study performed by Park *et al.* (2011) in the USA, they were reported to be 0.37 g kg^{-1} and 0.220 g kg^{-1}, respectively, during idling. Another example comes from China where the EF of BC was 0.160 g kg^{-1} for heavy diesel vehicles (Deng *et al.*, 2017). This clearly represents high variations in EFs of $PM_{2.5}$ and BC. However, the average EF of CO was found to be low compared to the study conducted by Park *et al.* (2011).

The emissions from diesel vehicles during idling are different from those when driving which are measured to obtain realistic data from diesel vehicles. There are various approaches to measure the EFs from these vehicles, like dynamometer and tunnel tests, which represent the different modes of driving conditions. However, dynamometer tests are quite expensive, while tunnel tests are limited to specific driving conditions. The present study was conducted for the first time in Nepal and in Kathmandu Valley to measure the EFs from diesel vehicles. We were able to measure the EFs in the idling condition as it was comparatively cheap, and this helped us obtain realistic EFs from the diesel vehicles at the local level. Although the EFs in the idling condition vary from those in the driving condition, still the data represents certain percentage of the latter. An earlier study observed that EFs of pollutants during the idling condition represents about 5 to 75% of the driving condition (McCormick *et al.*, 2000). Another study reported the representation of pollutants during the idling condition to be in the range of 66 to 74% of the driving condition (Park *et al.*, 2011). Although we were not able to perform the study during the driving condition, we can still compare the EFs obtained in the idling condition to those in the driving condition through these representative values. In this study, we compared idling EF with driving EF through the representative data given by Park *et al.* (2011). The EF of $PM_{2.5}$ in the idling condition calculated in this study was high compared to what the studies found in the USA, China, Germany and Switzerland during the driving condition, for both light and heavy diesel vehicles (Weingartner *et al.*, 1997; Kirchstetter *et al.*, 1999; Ban-Weiss *et al.*, 2008; Schneider *et al.*, 2008; Liu *et al.*, 2009; Wang *et al.*, 2011; Huo *et al.*, 2012) (Table 3). The EF of BC was also found to be higher compared to what the studies found in the USA,

Table 3. Comparison of EF (g kg^{-1}) with other EF studies.

References	Study type	Country	Year	Vehicle type	CO_2	CO	$PM_{2.5}$	BC
This study	Idling	Nepal	2017	Pickup (LDDV)	3,149	24	17.64	1.796
				Tipper (HDDV)	3,137	33	6.91	0.413
				Truck (HDDV)	3,101	56	37.09	1.282
				Bus (HDDV)	3,107	52	11.87	0.786
				Average (HDDV)	3,115	47	18.62	0.827
Park *et al*., 2011	Idling (different driving conditions)	USA	2007	HDDT	N/A	75	0.37	0.220
Subramanian *et al*., 2009	Dynamometer	Thailand	ca 2008	LDDT & HDDT	N/A	N/A	8.40	N/A
Deng *et al*., 2017	Idling	China	ca 2014	HDDT	N/A	N/A	N/A	0.160
Liu *et al*., 2009	On-board (driving)	China	2007 and 2008	LDDT	N/A	N/A	0.60	N/A
Huo *et al*., 2012	On-board (driving)	China	2007 and 2011	LDDT	N/A	32	1.86	N/A
				HDDT	N/A	29	1.37	N/A
Wang *et al*., 2011	On-board (driving)	China	2009	HDDT	N/A	42	2.35[a]	2.200
Kirchstetter *et al*., 1999	Tunnel (driving)	USA	1997	HDDV	N/A	N/A	2.50	1.300
Bishop *et al*., 2001	Remote sensing (driving)	USA	1997–1999	HDDT	N/A	31	N/A	N/A
Burgard *et al*., 2006	Remote sensing (driving)	USA	2005	HDDT	N/A	32	N/A	N/A
Ban-Weiss *et al*., 2008	Tunnel (driving)	USA	2006	HDDT	N/A	N/A	1.40	0.920
Weingartner *et al*., 1997	Tunnel (driving)	Switzerland	1993	LDDV	N/A	N/A	N/A	0.020
				HDDV	N/A	N/A	N/A	0.300
Schneider *et al*., 2008	On-board (driving)	Germany	2005	HDDT	3190	N/A	N/A	0.220

[a] represents $PM_{0.5}$.

LDDV represents light-duty diesel vehicle.

HDDV represents heavy-duty diesel vehicle.

LDDT represents light-duty diesel truck.

Germany and Switzerland during the driving condition, for both the light and heavy diesel vehicles (Weingartner *et al.*, 1997; Ban-Weiss *et al.*, 2008; Schneider *et al.*, 2008). However, the EF of BC observed in this study was also found to be lower compared to what the studies found in the USA in 1997 and China in 2009 for heavy diesel vehicles (Kirchstetter *et al.*, 1999; Wang *et al.*, 2011). The EF of CO for light- and heavy-duty vehicles was similar to what the studies found in the USA and China (Burgard *et al.*, 2006; Bishop *et al.*, 2001; Wang *et al.*, 2011; Huo *et al.*, 2012) (Table 3). These comparisons helped us to get a general idea about the EFs of diesel vehicles in the Kathmandu Valley with respect to those of other places in the world.

Of the three different pollutants, the variations in the EF of $PM_{2.5}$ were the most prominent. There are various reasons behind this apparent inconsistency in the emission data proffered by earlier studies conducted in different countries. They may include fuel composition; design and age of engine; manufacturing company; operating conditions; maintenance pattern; road conditions; traffic conditions; and environmental conditions (such as temperature, pressure, humidity, and altitude) during the measurements.

Influence of Vehicle Age on EF

The EFs of CO, $PM_{2.5}$ and BC were at their maximum for bus, truck, and pickup, respectively, during idling (Table 2). These three types of vehicle tested were older (more than 15 years) than the other vehicles tested in this study. The new vehicles (such as tippers and buses) had comparatively lower emissions than old trucks and buses. The EFs of $PM_{2.5}$, BC and CO for old trucks were more than fivefold, more than threefold, and one-threefold higher, respectively, than for new tippers. Similarly, the EFs of $PM_{2.5}$, BC and CO for old buses were twofold, one-fivefold, and two-fourfold higher, respectively, than for new buses. The study of heavy diesel vehicles conducted by Chen *et al.* (2007) indicated that in China, the EF of CO for old vehicles was threefold higher than for new vehicles. Another study conducted in Slovenia in 2011 disclosed the EF of BC for old, heavy diesel vehicles (of more than 10 years) to be 41% more than those for new vehicles (of 5–10 years) (Jezek *et al.*, 2015). Similarly, in China, PM emissions were higher in the case of old vehicles (China III emission standard) compared to the new ones (China IV emission standard) in 2016 (Wang *et al.*, 2018). The possible reason behind the high EF from old vehicles might have to do with poor engine combustion and irregular maintenance (Faiz *et al.*, 2006; Chen *et al.*, 2007).

Impact of Servicing on EF

The average EFs of $PM_{2.5}$ and BC reduced by 66% and 30%, respectively, just after vehicle servicing; however, the EF of CO increased by 2% during idling when averaged across four vehicles (Fig. 4). The EFs for both the conditions, before and after servicing, were higher in the beginning, and later on, decreased and attained a stage of stability (Figs. 5(a), 5(b) and 5(c)). The EF of $PM_{2.5}$ reduced by 57% and BC by 34% after servicing, particularly in the case of old buses. Similarly, the EFs of $PM_{2.5}$ and BC reduced by 65% and 15%, respectively, after new buses were serviced. One limitation in measurement was that there were no old, heavy cargo diesel vehicles for us to observe the impact of servicing. However, in the case of new tippers, we observed reductions in the EFs of $PM_{2.5}$ and BC by 46% and 1%, respectively, after servicing. In addition, servicing an old, light-duty vehicle (e.g., pickup) resulted in reductions in the EFs of $PM_{2.5}$ and BC by 81% and 34%, respectively. Surprisingly, there was an increase in the EF of CO in both heavy- and light-duty vehicles of any age (old and new) after servicing. A similar case was observed in a previous study conducted for gasoline vehicles (Stockwell *et al.*, 2016). The exact reason for this is still unknown (Stockwell *et al.*, 2016). Further detailed studies with a greater sample size are needed to understand this unusual increase in the EF of CO after servicing. However, the EF of CO decreased by 16% in the case of new tipper. Overall, it can be said that vehicle servicing helps to reduce emissions from different types of vehicles of any age, thereby contributing to the mitigation of diesel emissions in the Kathmandu Valley.

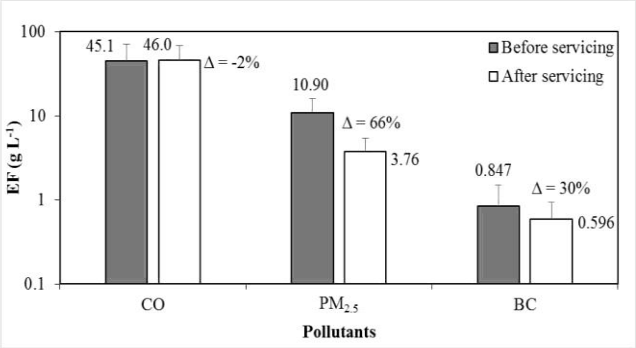

Fig. 4. Impact of servicing on EFs of CO, $PM_{2.5}$, and BC. Servicing included changing the engine oil, and replacing oil, diesel and air filters.

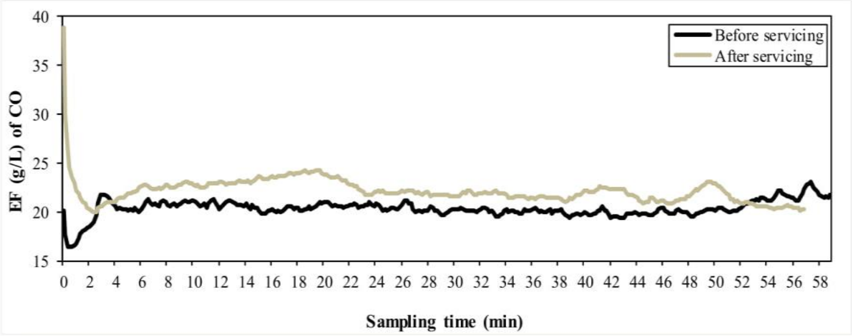

Fig. 5(a). Real-time EFs of CO for pick-up before and after servicing.

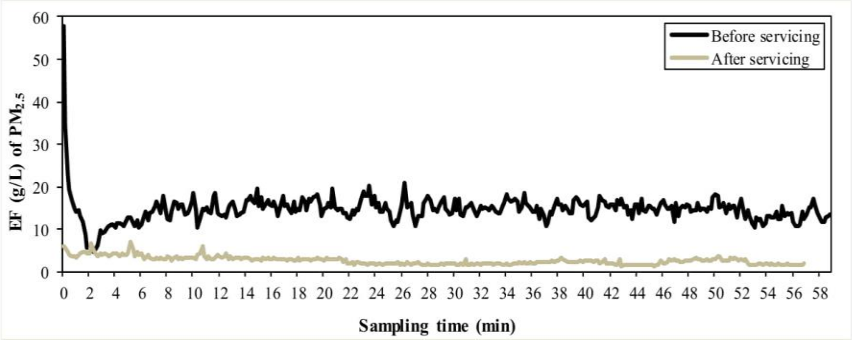

Fig. 5(b). Real-time EFs of $PM_{2.5}$ for pick-up before and after servicing.

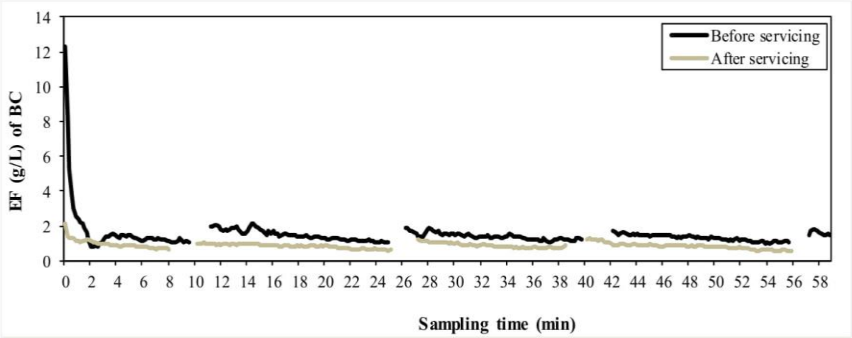

Fig. 5(c). Real-time EFs of BC for pick-up before and after servicing.

CONCLUSIONS

This is the first study of its kind in Nepal which provides the EFs of $PM_{2.5}$, BC and CO from diesel vehicles, while also exploring the impact of servicing on emissions. It presents roadside observations of diesel vehicles in the Kathmandu Valley, discusses the cost of general servicing, and provides measurements of the EF during idling condition from diesel vehicles. From these roadside observations, we found that only some specific vehicles in the entire diesel vehicle fleet were superemitters. We compared the EF during idling from this study with previous studies and found them to be higher than in other countries. The average total cost of general servicing for all types of vehicle was approximately USD 90–100. General vehicle servicing reduces the EF of $PM_{2.5}$ by 66% and of BC by 30% during idling. However, the study does not report an improvement in the EF of CO after servicing. The EFs presented here do not in themselves represent the entire diesel vehicle fleet of the Kathmandu Valley. However, the data provides a strong foundation for future research in the field of diesel vehicle emissions. Thus, this study helps in providing initial inputs toward the preparation of an emission inventory for diesel vehicles which can be used in quantifying global emissions, and it also provides a base to explore the cost-effectiveness of mandating routine servicing in order to mitigate vehicular pollution in the Kathmandu Valley.

There exists some limitations which we were not able to address in the present study. The roadside observations were not conducted during night-time due to safety issues. Similarly, the emissions from vehicles were not characterized based on speed of vehicles, seasonality of driving pattern as well as observations were not conducted during weekends. The EFs were also not measured during driving on plain or slope, and loading or unloading; further we were not able to carry out follow-up measurements after servicing due to strong resistance from the vehicle owners. Moreover, it was highly difficult to track down the same vehicle and get the owner's consent for undertaking emission measurements. In order to overcome such limitations, it is highly essential that academic, governmental and non-governmental agencies are brought together to organize larger campaigns in order to generate robust data on EFs from diesel vehicles.

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SUPPLEMENTARY MATERIAL

Supplementary data associated with this article can be found in the online version at http://www.aaqr.org.

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Research Article

A GLOBAL PERSPECTIVE OF VEHICULAR EMISSION CONTROL POLICY AND PRACTICES: AN INTERFACE WITH KATHMANDU VALLEY CASE, NEPAL

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ABSTRACT

Transport sector is growing most rapidly around the world in line with the urban and socio-economic growth, which is contributing to severe air pollution. Air pollution has been of much concern mainly due to air quality, human exposure, public health, climate change, and visibility reduction. At present, in the media and policy arena, significant attention is given to the transport air pollution and its effect. Although most of the developed countries established vehicular emission control practices, it is very primitive in the developing countries including Nepal. This paper highlights global policies/legislations that have been practiced for emissions control from high emitting vehicles based on the available literature. The insights and lessons based information presented in this paper will add value to the policy makers for creating strong policy packages of air quality management for Kathmandu valley including other parts of Nepal.

Keywords: Transport sector, Vehicular emissions, Air quality, Policy, Emissions control

INTRODUCTION

The world urban population has increased about fivefold in between 1950 to 2011 (i.e., passing from 0.75 billion to 3.6 billion) (UN 2012). Likewise, motor vehicle activity is growing rapidly (Aggarwal & Jain 2014, Badami 2004, Badami 2005). By 2040, world's population is expected to become nine billion. Therefore, the demand of energy is expected to grow by 30 percent from 2016 to 2040 (ExxonMobil 2013). Air quality around the world is found to be deteriorating and vehicular emissions is one of the main sources (Guo *et al.* 2016). The increase in vehicular emission is because of growing population, urbanization and socio-economic development (Wu *et al.* 2017, Guo *et al.* 2016). High fuel consumption from on-road vehicles emits greenhouse gases like carbon dioxide (CO_2), nitrous oxides (N_2O), methane (CH_4) and other harmful pollutants such as hydrocarbon (HC), particulate matters (PM), oxides of nitrogen (NO_x) and sulfur dioxide (SO_2) (Wu *et al.* 2017, Ribeiro *et al.* 2007, Kathuria 2004, Garg *et al.* 2006, Neeft *et al.* 1996, Raux 2004). Transport sector contributed 23 % of the world's CO_2 emissions and 74 % on-road CO_2 emissions in 2004 (Ribeiro *et al.* 2007). This data is closer to Chapman (2007), which reveals 26 % of the global CO_2 emissions. More emission prevails due to old and lack of adequate maintenance of road vehicles, high traffic congestion, fuel adulteration and poor road infrastructure. Although heavy-duty diesel vehicles (HDDV) represent lesser proportion, their emissions contribute significantly to air pollution problems (Kirchstetter 1999). Road vehicle emission also has been partly contributing for acid deposition, stratospheric ozone depletion and climate change (Shrestha 2018). The developed countries adopted strong legislation to reduce the automobile emissions and enhancing better air quality (Zhang *et al.* 1995). However, it is very primitive in the developing countries including Nepal. Vehicular emissions if uncontrolled may lead to increased level of health effects (e.g., respiratory disease, heart disease, cerebro-vascular disease, chronic obstructive pulmonary diseases, lower respiratory infections, and cancers) (Guttikunda *et al.* 2014, Neeft *et al.* 1996, Raux 2004) and can exacerbate local/regional warming because CO_2 and BC emissions have a high global warming potential (Das *et al.* 2018, Chapman 2007, Faiz 1993).

Due to rapid increase in vehicles registration (i.e., 13.8 % per year) at present (DoTM 2012), daily traffic number must have significantly increased. Therefore, it is reasonable to state that at present vehicles are the major sources of air pollution too in the Kathmandu valley. From 1989 until April 2015, 1,957,849 vehicles have been registered in Nepal. Out of which, 818,484 (42 %) vehicles was registered in Bagmati zone (DoTM 2015). Motorcycles occupies a large share (75 %) of Euro III technology, whereas, other vehicles (< 25 %) are Euro II or lower (Shrestha *et al.* 2013). The diesel fuel consumption in Kathmandu valley in 2012 was 15.46% of the total diesel consumption in Nepal (NRB 2012). This paper presents global policies and practices of vehicular emissions control. The lessons learned from other countries will add value to the policy review for better air quality management for Kathmandu valley and other parts of Nepal.

MATERIALS AND METHODS

Plethora of studies on policies context on vehicular emission control are available worldwide. The published literature of motor vehicles emission control in Asian countries (e.g., China, India and Nepal) as well as developed countries (e.g., United States and Europe) were thoroughly reviewed and incisively analyzed. International strategies and activities governments are undertaking to control emissions at present were deeply explored and compared among the countries. The national strategy and activities Government of Nepal currently being undertaken were also assessed. The policy gaps for Nepal were identified through papers review. Policy recommendations for vehicular emission control for Kathmandu valley were presented based on the scientific evidences, legislations, and practices adopted by developed and developing counties.

RESULTS AND DISCUSSION

Vehicle emission standard

China implemented emission standards and better fuel quality standards before 2000 to control vehicular emission. Later, the standards tightened in every 3 to 5 years. In just less than 13 years, China 1 (i.e., equivalent to Euro I) to China 5 (i.e., equivalent to Euro V) standard was formulated. To prevent from tampering behavior of the vehicle users, China amended the emission standard together with harsh penalties (Wu *et al.* 2017). Beijing was a pioneer in controlling the vehicular emissions (Wu *et al.* 2011). Realizing the vehicular emissions as one of the major sources and improving the air quality, China's state council was implemented air pollution prevention action plan in 2013 (SC 2013). The emission standard for HDDV for China 5 stage were 4.0 g/kWh CO, 0.55 g/kWh HC, 2.8 g/kWh NO_x, and 0.03 g/kWh PM (ICCT 2014). In India, government recently set up Bharat Stage (BS)-IV norms for vehicle emissions control. The maximum permissible limits for HDDV were 1.5 g/kmhr CO, 0.96 g/kmhr HC, 3.5 g/kmhr NO_x and 0.02 g/kmhr PM. However, BS III norms were 2.1 g/kmhr CO, 1.6 g/kmhr HC, 5.0 g/kmhr NO_x and 0.10 g/kmhr PM (CPCB 2017). Similarly, Europe strictly tightened the emission standard for heavy duty vehicles in the short span of time. The emission standards for NO_x for Euro I, Euro II, Euro III, Euro IV, Euro V and Euro VI were 8 g/km, 7 g/km, 5 g/km, 3.5 g/km, 2.0 g/km and 0.4 g/km respectively (Vestreng *et al.* 2009). Likewise, in United States, at present emission standards for heavy duty highway vehicles (Spark-Ignition) are 0.195-0.230 g/mi NMHC, 0.2-0.4 g/mi NO_x, 0.02 g/mi PM, and 7.3-8.1 g/mi CO depending upon the vehicle weight (USEPA 2016).

In Nepal, Environmental Protection Act-1997, National Ambient Air Quality Standard, NAAQS-2012 and Climate Change Policy-2011 were developed after realizing the need of better environment. However, those policies do not elaborate in-depth and clear picture for vehicular emissions control strategy. Nepal initiated activities for monitoring of exhaust emissions in the Kathmandu valley in 1993 and issuance of Green Sticker system was enforced since December 1999. Later, Nepal Vehicle Mass Emission Standard, NVMES-2000 was introduced and was again revised in 2012. For light duty diesel vehicles (LDDV) (< 2.5 tons), the upper permissible limit values (g/kWh) for passenger cars were 0.64 CO, 0.56 (HC+NO_x), 0.5 NO_x, and 0.05 PM. Similarly for LDDV (> 2.5 and < 3.5 tons), the permissible limit values (g/kWh) were 0.64-0.8 CO, 0.56-0.86 (HC+NO_x), 0.5-0.78 NO_x, and 0.05-0.1 PM. For diesel vehicles (> 3.5 tons), the upper permissible limit values (g/kWh) were 2.1 CO, 0.66 HC, 5.0 NO_x, and 0.10-0.13 PM (MoEST 2012). However, these tests have not been operated in Nepal yet. DoTM adopts smoke opacity test (SOT) for diesel vehicles in idling condition only. In general, the SOT for diesel vehicles is 2.5 per meter (naturally aspirated) and 3.0 per meter (turbo charged). According to Faiz *et al.* (2006), the smoke density for diesel vehicles ranges within 65-75 Hartridge Smoke Unit (HSU), as per the vehicle condition. The vehicle emission testing is only limited within Kathmandu valley and is applicable only to three and four wheelers.

Vehicle restriction

China has implemented an environmental labeling policy to mark high emitter vehicles. The yellow labels were marked for pre-China III HDDV. These vehicles were restricted to larger areas (e.g., Shanghai, Guangzhou, Shenzhen, and Hangzhou) after the release of the Action Plan in 2013 (Wu *et al.* 2016). In Europe, many countries (e.g., Austria, Denmark, Germany, Netherlands, and Sweden) have developed a low emission zone framework/legislation within the last decade (Holman *et al.* 2015). This is quite similar to yellow-labeled vehicles restricted to larger cities of China. Recently in Kathmandu valley, heavy duty vehicles were restricted to drive inside ring-road during the peak hours; however, the problem still persisted in the ring road as it lie close proximity to the main market centers.

Inspection and maintenance

China released a national regulation for inspecting and measuring on-road high emitting diesel vehicles using remote sensors (Ministry of Environmental Protection-China 2017). Vehicles inspection and maintenance practices have become successful in Gothenburg, Sweden too. The emission reduction performance was found better than Los Angeles, California and Melbourne (Zhang *et al.* 1995). DoTM is the sole organizations to conduct vehicle emission test in Nepal. Vehicle inspection is required once a year except commercial vehicles, which has provision to test every six months. To obtain a green sticker, it is crucial to pass the test and not allow the failed vehicles to operate in the Kathmandu valley. To pass the emission test, vehicle owner opt maintenance services. For road side emission check, only nominal penalty is charged for the failed vehicles. It was reported a decade ago that about 33-40 % of the diesel vehicles emissions can be reduced after proper maintenance (Faiz *et al.* 2006). However, there is lacks of vehicle maintenance policy in Nepal and the inspection as well as the emission test system has not been effectively implemented.

Fuel quality

China is currently promoting improved fuel quality as well as alternative fuels. The upper permissible limit of sulfur content in diesel vehicles was reduced from 2000 ppm (Euro 0) to 50 ppm (Euro IV) (Cai & Xie 2013, Cui *et al.* 2017). In India, SO_2 emissions have been found to be reduced greatly in the transport sector. This is due to the successive reduction of sulfur content [Bharat Stage (BS) III- 350 ppm and BS IV- 50 ppm] specification used in diesel vehicles imposed by fuel quality standard (Guttikunda *et al.* 2014, Amann *et al.* 2017). In Europe, in early 2000, maximum diesel sulfur permissible limit was 350 ppm. Later in 2005, the sulfur content was dropped to 50 ppm. The current guideline for diesel sulfur is 10 ppm (Euro VI) (ICCT 2016). Nepal Oil Corporation started supplying EURO III standard fuel since 2010. Maximum upper limit value of sulfur is 350 mg/kg of diesel (BS III) for Nepal.

Higher engine technology

China has made a progressive advancement in the Engine Technology from Euro III to Euro IV, switching towards electric vehicles, EV (Cai & Xie 2013, Cui *et al.* 2017) and Euro V vehicles in just short period of time. In India, policy has been developed to cut off vehicular emissions from HDDV. As per Supreme Court order, from 2002 onwards all buses were transformed to compressed natural gas, CNG in Delhi (Kathuria 2004). Later, the policy formulated two parallel standards such as BS IV for 11 major cities (Amann 2017) and BS III, for rest of the country. However, this policy did not address the consumers and businesses outside of the major cities equitably. All heavy duty trucks complied BS III standard only, and fuelling stations requiring low sulfur were also not accessible everywhere. Therefore, it is reasonable to say nothing prevents BS IV vehicles refueling in high-sulfur fuel area (ICCT 2013). Ministry of Road Transport and Highways of India decided to deploy Euro VI vehicles from 2020 onwards (Amann, 2017). In Europe, Euro VI has been implemented from 2014 to cutoff vehicular emissions (ICCT 2016). Realizing the issues of air pollution in the Kathmandu valley and to alleviate it, Government of Nepal banned the operation of highly polluting three wheelers diesel vehicles as well as import new two-stroke and second hand vehicles from 1999 onwards. Moreover, from 2000 onwards, government of Nepal promulgated to phase out 20 years old vehicles (Faiz *et al.* 2006). Those above factors have initiated scaling up of EV (e.g., Safa tempo) for public transportation. At present, all vehicles except HDDV imported to Nepal require compliance with EURO III emission standard.

Subsidy, tax exemptions and scrappage of high-emitting vehicles

To enhance EV sales, save energy and reduce vehicular pollution, subsidies and tax exemptions have been applied by the local and central Government of China. The scrappage program was launched to phase out yellow-labeled vehicles through proper subsidies/incentives. In this regard, Beijing has released a plan to cut off older vehicles (Wu *et al*. 2011). Similarly, Europe provides huge subsidies and greatly reduced tax to promote EV sales. In Norway, the excise tax reduction ranged from 39 % to 67 %. The Netherlands, France and UK excise tax reduction ranged from 10 % to 40 % (EC 2017). Decades ago, Nepal Government had provision to increase EV by waving custom duty and sales taxes. The policy makers and donor communities were interested to promote it in the Kathmandu valley (Dhakal 2003). Despite that, no strong initiative has been taken strategically and neither Government of Nepal has developed concrete plan of subsidies and reduction of excise tax to promote EV sales.

CONCLUSION

To minimize the vehicular air pollution, revision of emission standards of CO_2, NO_x, PM, CO, and HC of high emitting vehicles of Nepal are crucial. It should be based on the real- emission measurement work. Moreover, SOT should also be revised as mentioned above. In order to identify gross polluting vehicles in the Kathmandu valley, the application of remote sensing devices is crucial as in China. Using the device, traffic police can conduct on-road side emission test to get the real picture of vehicular performance. Likewise high emitting on-road vehicles should be seriously penalized. To assist in the identification of gross polluters in the Kathmandu valley, individuals can send SMS or text message to DoTM or traffic division from their mobile phones to report sightings of smoke belching vehicles. This approach was successful in Metro Manila. DoTM or traffic divisions may take action on vehicle owner for formal testing upon having such policy developed. Vehicle restriction strategy has not been effectively working in the Kathmandu valley. The problem of traffic jam and high emissions still persist around the main market centers in close proximity to the ring road. This issue must be addressed through study based policy. The vehicular inspection and maintenance policy can be one of the low cost options to reduce the emissions up to 40 % as per Faiz *et al.* (2006) findings. The inspection and maintenance policy should be developed for old model commercial vehicles as well. In order to maintain the fuel quality standard, timely inspection and fuel test of petrol pumps are essential. Moreover, technological change may play prodigious role in lowering the vehicular emissions. Government of Nepal should deploy advanced technology (e.g., Euro IV and higher models vehicles) as well as provide high subsidies and tax exemptions to promote EV sales. The scrappage of high emitting vehicles program can be other options for the Kathmandu valley through proper incentives from the government. To achieve better air quality or effective low carbon transport plan, development of comprehensive vehicular emission control polices and strategies are foremost steps for Nepal.

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