Environmental Processes
https://doi.org/10.1007/s40710-019-00367-w

ORIGINAL ARTICLE

Development of a Static Model to Identify Best Management Practices for Trace Metals from Non-Exhaust Traffic Emissions

Matthew Fiala[1] · Hyun-Min Hwang[1]

Received: 12 October 2018 / Accepted: 17 March 2019/Published online: 28 March 2019

Abstract

Risk through exposure to non-exhaust traffic emissions continues to increase as both the number of vehicles and traveling distances continue to increase with global urbanization. To better understand their impacts on the urban environment, a contaminant pathway was developed to describe important release mechanisms, transitory environmental media, and exposure media. Sources of contaminants were identified and characterized using published literature values. Concentrations of non-exhaust sources were used in conjunction with mean emission factors to estimate contaminant loads from individual vehicles (μg/km/veh). Published daily vehicle distances traveled were used to estimate total annual emissions (kg/yr) for the United States and Houston metropolitan area. This equates to approximately 5.1 million kg of Cu, 12.8 million kg of Zn, 4.9 million kg of Pb, and 2400 kg of Cd being released each year in the United States. Tires are responsible for 92% of total Zn emissions, with heavy-duty vehicles responsible for 77% of these emissions. Tires are also responsible for 86% of total Cd emissions. Brake dust contributes to 99.9% of Cu emissions. Wheel weights contribute approximately 94% of total Pb emissions. Identified best management practices include: 1. installation of grass buffer zones (e.g., rain gardens, vegetated swales) immediately adjacent to road surfaces, and 2. permeable pavements and green roofs near major highways. There are several limitations, assumptions and uncertainties associated with the study due to its static nature. However, evidence is substantial for the need to create new policies that address the pollution created by non-exhaust traffic emissions.

Keywords Trace metals · Environmental contamination · Transportation · Model

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s40710-019-00367-w) contains supplementary material, which is available to authorized users.

✉ Matthew Fiala
matthew.fiala@knoxmedical.com

[1] Department of Environmental and Interdisciplinary Sciences, Texas Southern University, 3100 Cleburne, Houston, TX 77004, USA

1 Introduction

Operation of motor vehicles is a major source of environmental contaminants, especially in urban areas (Hwang et al. 2016, 2018). To date, non-exhaust emissions have been relatively unaddressed by government agencies and the automotive industry, whereas strict exhaust emission controls have led to enormous improvements in exhaust emissions (Kukutschová and Filip 2018). Non-exhaust emissions include brake dust, tire wear, and pavement wear which are generated through abrasive forces, emitting particulate matter (PM) (Adachi and Tainosho 2004). Bound to released PM are toxic trace metals, including lead (Pb), copper (Cu), zinc (Zn), cadmium (Cd), chromium (Cr), cobalt (Co), nickel (Ni), and vanadium (V).

Aerosolized particles eventually deposit on pervious and impervious surfaces. When deposited on pervious surfaces, trace metals infiltrate into the soil column (Fiala 2017). Particles deposited on impervious surfaces are mobilized via stormwater runoff and transported to receiving surface waters that cause toxic effects to sensitive aquatic organisms (Hwang et al. 2016). Finer particles that deposit on or near road surfaces become easily re-suspended into the air by wind or traffic-related forces (Acosta et al. 2011), and can enter indoor environments (Kuo et al. 2012). High trace metal concentrations in outdoor soil and air are good indicators that indoor air and dust is significantly contaminated (Campbell et al. 2018). Trace metals that enter a home is a significant source of exposure via inhalation or ingestion (Gulson and Taylor 2017).

Lead, potentially the most toxic non-exhaust emission metal, is a non-threshold developmental neurotoxicant without any safe levels of consumption. Toxic effects are well documented in response to historical exposure (lead-based paint and automotive gasoline). High blood-levels cause damage to internal organs and central nervous system (Hammond and Dietrich 1990), with lower exposure responsible for learning disabilities in young children (Needleman et al. 1990; Wang et al. 2009). Although Zn and Cu are considerably less toxic, adverse health effects associated with excessive Zn exposure include immune and hematologic complications, with excessive Cu exposure resulting in gastrointestinal, hepatic and renal effects (US EPA N.D.).

Once Pb and other metals enter the environment, they are continuously transported between air, water and soil by natural chemical and physical processes (e.g., weathering, precipitation, dry deposition) and have a potential for human and environmental exposures that continue for generations (ATSDR 2007). Copper in urban streams frequently exceeds probable effect concentrations for predicting sediment toxicity. Copper concentrations in stream sediments are expected to continue increasing unless emissions from major sources such as brake pads are regulated (Hwang et al. 2016).

Brake dust and tire wear are two widely studied sources due to associated adverse effects on fish and other aquatic organisms resulting from Cu and Zn toxicity (Hwang et al. 2016). Lesser known non-exhaust sources include wheel balancing weights, motor oil leaks, and pavement wear. Affixed to nonpermanent locations, lead wheel weights inevitably become loose due to wheel jarring, dislodge, and are pulverized into small particles by vehicle traffic (Root 2000). Motor oil that leaks on pavement surfaces contain a variety of trace metals related to the oil formulation and engine wear with the water-soluble fraction transported by stormwater runoff (Kennedy et al. 2002). Concrete and asphalt pavements contain highly variable trace metal concentrations based on the local supply of aggregates, binders and fillers (Thorpe and Harrison 2008). Particulate matter and trace metals from pavements are released into the environment by abrasive forces with vehicle tires.

Risk associated with non-exhaust traffic emissions continue to increase as the number of vehicles and traveling distances increase with global urbanization. A better understanding of non-exhaust sources, the transport of emitted metals, and research gaps is necessary to implement evidence-based best management practices that reduce risk associated with environmental and human exposures.

2 Methods

A conceptual transport pathway of contaminants from non-exhaust emissions was developed to describe release mechanisms, transitory environmental media, and biological exposure media in the urban environment (Fig. 1). Sources of contaminants were identified and characterized using published literature values. Source trace metal concentrations (Table 1) were used in conjunction with mean emission factors (Table 2) to estimate contaminant loads for individual vehicles (μg/km/veh). The study focuses on metals zinc (Zn), copper (Cu), lead (Pb), and cadmium (Cd) because their concentrations in non-exhaust emissions are well-documented compared to other metals. These metals are also commonly reported tracers for environmental contamination by brake dust and tire wear (Padoan and Amato 2018; Penkala et al. 2018).

Trace metal concentration of each source (Table 1) were derived by compiling concentrations from literature (see Supplemental Information) to calculate an average concentration. Limited literature is published regarding the precise chemical composition of bulk road surface materials in asphalt and concrete pavement, likely due to the many ways of choosing the composition and proportion of materials used to make a concrete mix (Penkala et al., 2018). It is assumed that study areas are composed of common pavement mixtures reported by the U.S. Federal Highway Administration, with common concrete mixtures being composed of approximately 70% Portland cement, 15% fly ash, and 15% other pozzolanic material (FHWA 1998). It is assumed that trace metal concentrations in the pozzolanic material is negligible compared to concentrations in fly ash and Portland cement. Published concentrations for fly

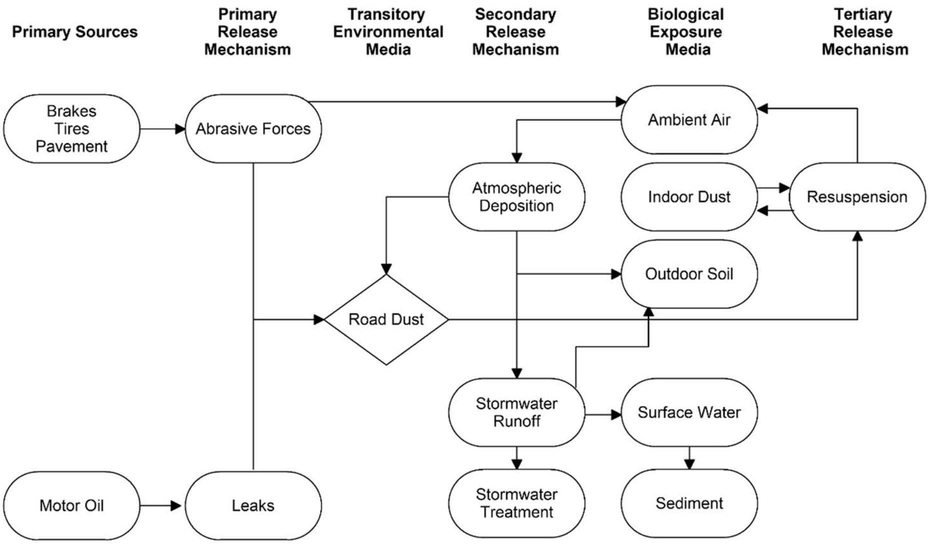

Fig. 1 Conceptual transport pathway of contaminants from non-exhaust emissions

Table 1 Trace metal concentrations for each source, C_i (μg/g)

Non-exhaust emission source	Zn	Cu	Pb	Cd	Reference
Brake Dust	12,290	66,380	3502	3.9	Supplemental Material
Tire Wear	11,015	3.17	4.51	1.33	Supplemental Material
Asphalt Pavement	70	35	7	0.1	Supplemental Material
Concrete Pavement	39	23	63	3.34	Supplemental Material
Used Oil (μg/mL)	125	2.1	1.1	0.1	Davis et al. 2001
Wheel Balancing Weights	0	0	950,000	0	Root 2000

ash ($N = 5$) and Portland cement ($N = 2$) is compiled (see Supplemental Information), and an approximate average concentration for concrete pavement was determined using mass-weighted average of fly ash and Portland cement. Published concentrations of asphalt pavement ($N = 2$), bitumen ($N = 3$), and aggregates ($N = 3$) were compiled (see Supplemental Information); assuming a common composition of 95% aggregates and 5% bitumen (Thorpe and Harrison 2008), an approximate average concentration for asphalt pavement was determined.

Total non-exhaust emissions for each vehicle type (E_v) was calculated using the below formula, where C_i is concentration of a metal in each source, i; EF_i is emission factor for total suspended particles (TSP) for each source, i; DT_v is average distance traveled for each vehicle type, v; N is number of each vehicle type in study area, and P is the number of sources.

$$E_v = \sum_{j=1}^{p} C_i \, EF_i \, DT_v \, N$$

When emission factors are unavailable for TSP, emission factors for PM_{10} were used in conjunction with the typical size profiles reported by the European Environmental Agency to approximate TSP (EEA 2003), although it is known that size fractions of emitted particles are highly variable. The static variables used in this study provide simplicity and proof of

Table 2 Mean Emission factor for each source, EMF_i (mg/km/veh)

Type	Output (TSP)
Brake Wear	
Passenger	12.5 mg/veh/km
Mid-Duty	29 mg/veh/km
Heavy Duty	55 mg/veh/km
Tire Wear	
Passenger	29 mg/veh/km
Mid-Duty	652 mg/veh/km
Heavy Duty	2934 mg/veh/km
Pavement Wear	
Passenger	11 mg/veh/km
Mid Duty	43 mg/veh/km
Heavy Duty	74 mg/veh/km
Leaked Motor Oil	
Passenger	2.8 ul
Mid/Heavy-Duty	2.1 ul
Wheel Weights	
Fall off rate	0.97 mg/veh/km
Fraction pulverized per day	0.0272

TSP, Total suspended solids; literature review of emission factors is shown in the Supplemental Material concept, while more dynamic variables are needed to increase accuracy and precision by representing the natural variation that occurs between individual vehicles.

Total annual emissions were calculated for the Houston metropolitan area using daily vehicle km traveled from 09/1/2017 to 08/31/2018 for Houston District, Texas, USA (TxDOT 2018), and for the United States using average calculated vehicle/km/yr traveled shown in Table 3 (US DOT 2018). To approximate emissions from passenger, mid-duty and heavy-duty vehicles, it is assumed that the vehicle fleet in Houston metropolitan is the same composition as the United States vehicle fleet shown in Table 3. In the Houston metropolitan area, 152 million km were driven daily from 09/1/2017 to 8/31/2018 (TxDOT 2018), with light-duty (passenger), mid-duty, and heavy-duty vehicles traveling approximately 138 million, 6 million, and 9 million km, respectively. Because no accurate information exists regarding the composition of Houston's road surfaces, it is assumed that the ratio of asphalt to concrete roads are the same as their occurrence in the United States, with approximately 94% of all paved road surfaces being asphalt-based (FHWA 2015), and most of the remaining surfaces being composed primarily of concrete pavement.

3 Results and Discussion

Trace metal emission rates (μg/km/veh) from passenger, mid-duty and heavy-duty vehicles are presented in Table 4. Non-exhaust emission rates are normalized using national vehicle miles traveled for each vehicle type to estimate annual emissions for the United States (Table 5). Tires are the primary source of Zn, responsible for 92% of total Zn emissions, with heavy-duty vehicles responsible for 77% of these emissions. Brake dust contributes to 99.9% of Cu emissions. Fallen wheel weights contribute approximately 94% of total Pb emissions. Used motor oil and pavement wear are comparatively insignificant sources of trace metals.

Approximately 5.1 million kg of Cu, 12.8 million kg of Zn, and 4.9 million kg of Pb is released each year in the United States through brake dust, tire wear, and fallen wheel weights, respectively. Lead is significantly over-estimated because it does not consider the banning of wheel weights in individual states (i.e., California, Washington) and local governments. Approximately, 126,000 kg of wheel weights is pulverized annually in a situation with no current best management practices such as sweeping and washing of pavement surfaces. Previous studies estimate 1.42–4.8 million kg of Cu, 8–11 million kg of Zn, and 0.43–1.8 million kg of Pb is released each year in the United States from brake dust, tire wear, and wheel weights, respectively (Hwang et al. 2016). Compared to previous studies, trace metal

Table 3 Average distance traveled (DT_v) and number (N) of each vehicle type in the U.S.

Vehicle type	Number of vehicles [a]	Vehicle-km/yr (Million) [b]	Average (km/veh/yr)	Percent of total km driven
Passenger	247,644,982	4,586,176	18,519	91%
Mid-Duty	8,746,882	182,400	20,853	4%
Heavy-Duty	2,752,043	280,922	102,077	6%

US DOT (2018)

[a] Table 1–11: Number of U.S. Aircraft, Vehicles, Vessels, and Other Conveyances

[b] Table 1–35: U.S. Vehicle-Miles (Millions)

Table 4 Trace metal emission rates from non-exhaust sources (μg/km/veh)

Non-exhaust Emission Source	Zn	Cu	Pb	Cd
Brake Dust				
Light Duty	154	830	44	0.05
Mid Duty	356	1925	102	0.11
Heavy Duty	676	3651	193	0.21
Tire Wear				
Light Duty	319	0.09	0.13	0.04
Mid Duty	7182	2.1	2.9	0.87
Heavy Duty	32,318	9.3	13	3.9
Pavement Wear				
Light Duty	0.75	0.38	0.1	0.003
Mid Duty	2.9	1.5	0.4	0.01
Heavy Duty	5.0	2.5	0.8	0.02
Motor Oil				
Light Duty	0.35	0.006	0.003	0.0003
Mid Duty	0.26	0.004	0.002	0.0002
Heavy Duty	0.26	0.004	0.002	0.0002
Wheel Weights				
All Vehicles			922	
Pulverized			25	

Table 5 Trace metal emissions from non-exhaust sources for the United States (kg/year)

Emission Source	Zn	Cu	Pb	Cd
Brake Dust				
Light Duty	704,545	3,805,348	200,758	224
Mid Duty	65,009	351,121	18,524	21
Heavy Duty	189,888	1,025,612	54,108	60
Subtotal	959,442	5,182,081	273,390	304
Tire Wear				
Light Duty	1,464,973	422	600	177
Mid Duty	1,309,948	377	536	832
Heavy Duty	9,078,785	2613	3717	1096
Subtotal	11,853,705	3411	4853	2106
Pavement Wear				
Light Duty	3437		523	15
Mid Duty	534	269	81	2
Heavy Duty	1417	713	215	6
Subtotal	5388	2711	819	23
Motor Oil				
Light Duty	1605	27	14	1
Mid Duty	48	1	0	0
Heavy Duty	74	1	1	0
Subtotal	1727	29	15	1
Wheel Weights				
Light Duty			4,226,126	
Mid Duty			168,080	
Heavy Duty			258,868	
Subtotal			4,653,074	
Pulverized			126,564	
Total	12,820,263	5,188,232	4,932,152	2435

emissions of Cu and Zn exceed previous studies, although the calculated value may be accurate due to an increase of vehicle numbers and travel distances in recent years.

An estimated 149,000 kg of Zn, 58,000 kg of Cu, 54,000 kg of Pb, and 20 kg of Cd is released into Houston each year. Lead emissions of pulverized wheel weights (125,000 kg/yr) is overestimated because road sweeping and washing likely occurred within Houston Metropolitan throughout the year; however, the frequency of cleanings was not within the scope of this study. There is a moderate level of confidence in the accuracy of Zn and Cu results since the method was verified using comparisons with previous studies on the national scale.

By describing contaminant loads associated with the sources, the influence of best management practices and environmental policies on loads can be estimated. The developed static transport model (Fig. 1) does not account for spatial and temporal variability of meteorological conditions such as wind intensity and direction, and rainfall intensity and duration, which inevitably affect the accumulation of particulate pollutants and subsequent mobilization (Davis and Birch 2010). There are too many uncertainties and unknown variables to accurately predict their fate and transport in the urban environment as detailed in Fig. 1.

Transport of non-exhaust emissions can be broadly predicted using available data from Steiner et al. (2007). Approximately 36–65% of trace metals are transported through highway runoff and spray, 35–64% of trace metals are dispersed through drift and deposit within 300 m from the road, and 18% of the total amount dispersed through drift is deposited within 5 m (Fig. 2). Given that most trace metals in highway runoff are adhered to coarse particulates (Furumai et al. 2002), most of the metals that become mobilized by highway runoff are likely to settle close to stormwater outfalls, while metals bound to particles <20 μm will remain suspended in receiving streams if not appropriately treated prior to discharge. Installation of grass buffer zones (e.g., rain gardens, vegetated swales) immediately adjacent to road surfaces will likely sequester loads deposited within 5 m, thus reducing total trace metal loads entering stormwater and receiving streams. In the Houston Metropolitan area, this equals approximately 27, 10, and 10 t/yr of Zn, Cu and Pb, respectively. Grass buffer zones will likely capture the fraction of metals that is dispersed through spray when vehicles hit patches of water on road

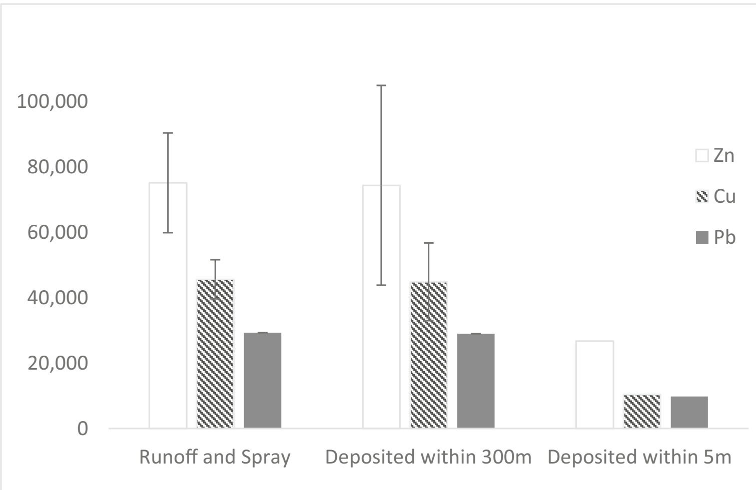

Fig. 2 Estimated contribution of contaminants from non-exhaust emissions in the Houston Metropolitan area transported through highway runoff and spray, atmospheric deposition within 300 m, and atmospheric deposition within 5 m of road surfaces (kg/yr)

surfaces, and the fraction of highway runoff that passes through grass buffer zones prior to entering storm drains. Implementation of green roofs and porous pavement can capture a significant portion of the remaining aerosolized particles that deposit within 300 m or further from road surfaces. Implementation of all these BMPs simultaneously will be cost prohibitive; however, municipalities can begin by implementing grass buffer zones which are the most cost-efficient BMP, following by permeable pavement and green roofs.

Resuspension of accumulated road dust is an important tertiary release mechanism (Fig. 1) that can be minimized by hardening unpaved roads with chemical-binders as well as water-spraying in dry periods. Frequent washing and sweeping of paved roads are a practical best management practice for paved roads. Reduction of particulate matter emissions up to 90% have been achieved by frequent washing and sweeping, limiting vehicle speed and mass, and by using a noise barrier on roads (Penkała et al. 2018). A recent study has shown that the physical removal of trace metals from road surfaces through washing and sweeping does not sufficiently limit human exposure to PM from non-exhaust emissions (Harrison and Hester 2017). A list of other studies indicates that PM close to roads may be much more toxic or have higher carcinogenic potential than PM outside these areas (Penkała et al. 2018).

Inferences can be made based on the contribution of major sources to trace metal loads in the Houston Metropolitan area (Fig. 3). Lead wheel balancing weights contribute significant amounts of Pb in urban environments in geographic areas which have not banned their use. To prevent the accumulation of lead in urban environments, it is necessary for federal agencies to enact policies banning lead from wheel balancing weights. The European Union, and some US states have already banned the use of Pb wheel weights, requiring the use of non-toxic materials. Steel, Zn, and composite materials are excellent alternatives to Pb and have already been implemented in Washington (Washington State Department of Ecology N.D.). Lead wheel weights also dominate the market in Canada; approximately 110–131 t of lead in detached wheel weights are pulverized on Canadian roadways each year. A phase out of lead wheel weights is expected to result in a decrease of pre-school children's lead blood levels up to 0.4 μg/dL (Campbell et al. 2018). In Europe, lead wheel weights were banned on new vehicles and after-market wheels as of July 1, 2005 in response to concerns about losses on roadways and inappropriate disposal by tire retailers and scrap processors (European Commission 2000).

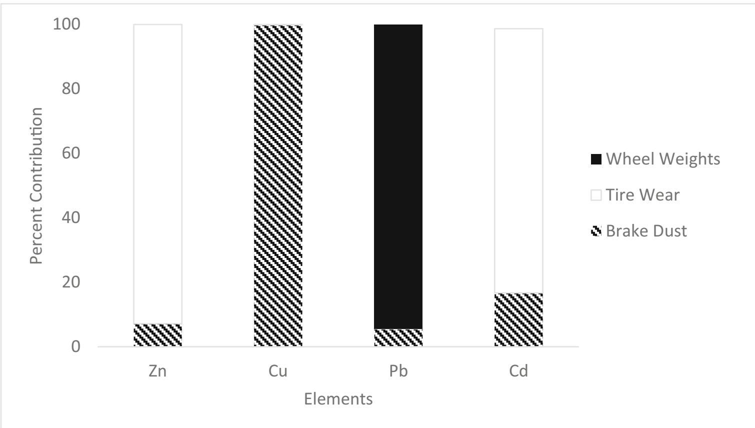

Fig. 3 Percent contribution (%) of major non-exhaust traffic sources to annual trace metal loads in the Houston Metropolitan area

As shown in Fig. 3, trace metal contribution from pavement and leaked motor oil is negligible compared to brake dust, tire wear and wheel weight contribution. This supports the conclusion of Penkała et al. (2018) that direct road surface abrasion is of minor importance when the road is undamaged and that PM emissions are significantly lower than emissions from other sources such as brakes and tires. Extensive research is needed to establish reliable emission factors and chemical composition of pavement road wear in laboratory-controlled studies to quantify the influence of vehicle type, speed and environmental conditions (Penkała et al. 2018). The contribution of pavement wear is currently uncharacterized as a contributing source of PM near roads and its influence on the environment is unknown requiring further evaluation. To effectively manage Cu pollution, environmental policies must address the growing concern of environmental damage caused by brake pads with high concentrations of copper. Either Cu must be removed from brake pad formulations, or advancements are needed in the design of modern automobile braking systems. Modern open-design of brakes on passenger vehicles diffuse brake dust immediately into the surrounding environment, while the closed-design of disc brakes contain a fraction of generated brake dust. Research is also needed to investigate new formulations for tires to limit the use of Zn and other toxic chemicals.

3.1 Limitations, Assumptions and Uncertainties

There are several factors limiting accuracy and precision when describing the sources and transport of non-exhaust traffic emission loads entering environmental matrices. Emission rates and size distribution of non-exhaust emissions have a high variability between individual vehicles and environmental conditions. There is a huge knowledge gap on spatial variability of non-exhaust emissions and the subsequent development of road dust and its resuspension. Road dust is composed of non-exhaust vehicle emissions in addition to soil and particle elements from other sources that have settled on the pavement surface. Several factors are responsible for road dust buildup and its mobilization in the environment including pavement characteristics, traffic intensity and speed, fleet composition, proximity to traffic lights, and meteorological conditions. Most aerosolized particulate mass from non-exhaust emissions, regardless of the traffic type, is made of resuspended dust (Padoan et al. 2017). Due to the large study areas and high level of uncertainty, the researchers were unable to predict trace metal loads as it moves between soil, sediment, air, and indoor matrices. Future studies are needed to address the natural variation of trace metal concentrations in sources, and their dynamic emission factors that vary considerably with many different variables.

3.1.1 Tire and Pavement Wear

The type of vehicle tires and travel speed have drastic effect on pavement wear rates, with studded winter tires having significantly greater PM emissions compared to non-studded winter tires, and summer tires were shown to have very little effect on PM production in comparison (Gustafsson et al. 2009). Dissimilar aggregate materials in pavement result in significant differences in PM emissions, with granite aggregates resulting in 70% higher emissions than a quartzite pavement of the same aggregate size (Gustafsson et al. 2009), and larger aggregate size resulting in lower emissions (Amato et al. 2013). Pavement surface composition and chemistry is known to change drastically over very short distances, which is a truly problematic issue for modeling and source apportionment studies. Pavement wear is the most difficult source to trace because the high variability of geological material used prevents the identification of unique elemental tracers (Padoan and Amato 2018; Gustafsson 2018).

Tire wear particles, generated through abrasion with pavement surfaces, is also known to vary significantly in size under dissimilar environmental conditions including: pavement roughness, severity of road alignment, rapid acceleration and deceleration, vehicle loading, and tire properties (Bennett and Greenwood 2001). Much of the tire emission data does not use a consistent marker for measuring tire wear particles, and differences are likely explained by varied methods used to calculate emission rates such as derivation from emission inventories, receptor modeling, and statistical models using source profiles, as well as direct measurements in simulated laboratory experiments and roadside air sampling. The emission rates used are from the most recent studies; there are many studies conducted 10–20 years ago that should be used with caution as tire wear rates have improved with advanced tire technologies (Panko et al. 2018).

Given the variety of factors that influence the generation of tire and pavement wear particles, future efforts at characterizing this source need to consider not only the characteristics of the tire, but also the vehicle to which it is mounted, the pavement the vehicle is driven, and the design of roadway systems that affect the manner in which the vehicle is operated (Panko et al. 2018).

3.1.2 Brake Wear

Emission factors and metal concentration of brake dust is highly variable between individual brake pads because of several thousand different raw materials used in formulations of brake pads by different brands (Filip et al. 1997). Each combination of speed, temperature, and pressure of the brake pads result in a different amount of wear. Less severe braking applications at lower temperatures generally result in lower wear and friction debris (Kukutschová and Filip 2018). At high-temperature conditions, adhesive, abrasive, fatigue, and oxidative wear are observed, while “mechanical” wear dominates at lower temperatures. Oxidative wear can generate very-fine (submicron-sized) particles, whereas mechanical wear (abrasive and fatigue wear) typically release larger particles in the PM_{10} and $PM_{2.5}$ fractions. The stochastic nature of the braking process and complex mechanochemical interactions during the braking process make it difficult to predict chemistry and particle size. Produced break wear debris is a complex mixture of particles ranging in size from several nanometers to millimeters, and chemistry that is significantly different from the original pad material constituents (Kukutschová et al. 2011; Kukutschová and Filip 2018). Although it is not possible to simulate all braking scenarios, researchers and manufacturers use a variety of tests, including small laboratory tests (pin-on-disc testing), subscale and full-scale dynamometers and real field tests to demonstrate how tested brakes perform under a wide range of conditions (Lee and Filip 2013).

Earlier studies report airborne fraction of brake wear between 30 and 50% (Garg et al. 2000; Sanders et al. 2003). The mass of airborne emissions increases with temperatures exhibiting maximum values at 400C (Alemani et al. 2016), however, the percentage of airborne PM was higher for 100C than for 400C tests in a brake dynamometer. The wind tunnel experiment performed by Sanders et al. (2003) is more representative of airflow around the brake hardware in real vehicle conditions. Dynamometer studies are not well representative of this occurrence, while it is difficult to clearly distinguish direct brake wear from other sources in field studies. Airborne brake wear particles have also been detected during acceleration events (Hagino et al. 2016), suggesting that resuspension of wear particle should be included in brake emission measurements.

The wide range of emission factors between brake wear studies is a result of several variables (testing device, testing procedure, sampling conditions, brake materials, ect.).

Currently, there are no recommended approaches for the generation, measurement, and expression of brake wear emissions. Further studies are needed to develop a representative testing methodology reflecting the contribution of brake materials to environmental pollution (Kukutschová and Filip 2018).

3.1.3 Wheel Weights

The contribution of lead wheel weights to non-exhaust emissions through falling on to road surfaces and their subsequent pulverization by vehicular traffic was estimated using the approach by Root (2000). A steady state of lead wheel weights was assumed with the rate of pulverized Pb assumed to remain constant. In real-world situations, there will be periodic decreases in the lead wheel weights on road surfaces due to cleaning activities that result in their removal from the roadway. The depletion of lead wheel weights on roadsides was not considered in the current study. Geographical regions such as the European Union, have already banned lead wheel weights, and will have significantly lower contribution of lead. In such cases, lead wheel weights should be removed as a source from the static model.

4 Conclusion

Calculated non-exhaust traffic emissions for Zn and Cu in the United States are slightly higher than previous studies that estimated 8–11 million kg of Zn, 1.42–4.8 million kg of Cu, and 0.43–1.8 million kg of Pb released each year in the United States (Hwang et al. 2016). Tires are identified as the primary source of Zn, responsible for 92% of total Zn emissions, with heavy-duty vehicles responsible for 77% of these emissions. Brake dust contributes to 99.9% of Cu emissions. Fallen wheel weights contribute approximately 94% of total Pb emissions. Used motor oil and pavement wear are comparatively insignificant sources of trace metals. Lead contamination through wheel weights are likely overestimated because road surface sweeping and washing throughout the year were not considered, in addition to banned use in individual states (i.e. California, Washington) and local governments.

The contaminant pathway was used to identify the installation of grass buffer zones (e.g., rain gardens, vegetated swales) immediately adjacent to road surfaces as the most efficient method to sequester loads associated with non-exhaust emissions. Installation of permeable pavement and green roofs are additional measures that can act as a sink for trace metals that deposit further away from road surfaces. Although many limitations, assumptions and uncertainties are associated with the study due to its static nature, evidence is substantial for the need to create new policies that address the pollution created by non-exhaust traffic emissions.

References

Acosta JA, Faz Á, Kalbitz K, Jansen B, Martínez-Martínez S (2011) Heavy metal concentrations in particle size fractions from street dust of Murcia (Spain) as the basis for risk assessment. J Environ Monit 13(11):3087–3096

Adachi K, Tainosho Y (2004) Characterization of heavy metal particles embedded in tire dust. Environ Int 30(8): 1009–1017

Alemani M, Nosko O, Metinoz I, Olofsson U (2016) A study on emission of airborne wear particles from car brake friction pairs. SAE International Journal of Materials and Manufacturing 1:147e157. https://doi.org/10.4271/2015-01-2665

Amato F, Pandolfi M, Alastuey A, Lozano A, González JC, Querol X (2013) Impact of traffic intensity and pavement aggregate size on road dust particles loading. Atmos Environ 77:711–717
ATSDR U (2007) Toxicological profile for lead. US Department of Health and Human Services, public health service, Agency for Toxic Substances and Disease Registry. In: Atlanta (GA) available from www.atsdr.cdc.gov/toxprofiles/tp.asp?id=96&tid=22
Bennett CR, Greenwood ID (2001) Volume seven: modelling road user and environmental effects in HDM-4. The highway development and management series. Report to the International Study of Highway Development and Management Tools, University of Birmingham
Campbell PM, Corneau E, Nishimura D, Teng E, Ekoualla D (2018) Cost-benefit analysis for a lead wheel weight phase-out in Canada. Sci Total Environ 637:79–90
Davis AP, Shokouhian M, Ni S (2001) Loading estimates of lead, copper, cadmium, and zinc in urban runoff from specific sources. Chemosphere 44(5):997–1009
Davis B, Birch G (2010) Comparison of heavy metal loads in stormwater runoff from major and minor urban roads using pollutant yield rating curves. Environ Pollut 158(8):2541–2545
EEA (2003) Atmospheric emission inventory guidebook, 3rd edition. Group 7: road transport. European Environmental Agency. http://www.eea.europa.eu/publications/EMEPCORINAIR5/page016.html. Accessed 19 Mar 2019
European Commission, 2000. EC directive 2000/53/EC. The "ELV Directive" 18th September 2000. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CONSLEG:2000L0053:20050701:EN:PDF
Fiala MJ (2017) Development of a transport model for trace metals from non-exhaust traffic emissions. Doctoral dissertation, Texas Southern University
Filip, P., Kovarik, L., & Wright, M. A. (1997). *Automotive brake lining characterization* (no. 973024). SAE Technical Paper
FHWA. (2015). Public road length: miles by type of surface and ownership/functional system, national summary. Federal Highway Administration (FHWA). United States Department of Transportation. https://www.fhwa.dot.gov/policyinformation/statistics/2015/hm12.cfm
FHWA. (1998). Materials notebook. Chapter 5: Portland cement concrete. Federal Highway Administration. https://www.fhwa.dot.gov/pavement/materials/matnote.cfm
Furumai H, Balmer H, Boller M (2002) Dynamic behavior of suspended pollutants and particle size distribution in highway runoff. Water Sci Technol 46(11–12):413–418
Garg BD, Cadle SH, Mulawa PA, Groblicki PJ (2000) Brake wear particulate matter emissions. Environmental Science & Technology 34(21):4463–4469
Gulson B, Taylor A (2017) A simple lead dust fall method predicts children's blood lead level: new evidence from Australia. Environ Res 159:76–81
Gustafsson M, Blomqvist G, Gudmundsson A, Dahl A, Jonsson P, Swietlicki E (2009) Factors influencing PM_{10} emissions from road pavement wear. Atmos Environ 43(31):4699–4702
Gustafsson M (2018) Review of road wear emissions: a review of road emission measurement studies: identification of gaps and future needs. Non-Exhaust Emissions, p 161–181
Hagino H, Oyama M, Sasaki S (2016) Laboratory testing of airborne break wear particle emissions using a dynamometer system under urban city driving conditions. Atmos Environ 131:269–278
Hammond PB, Dietrich KN (1990) Lead exposure in early life: health consequences. Reviews of Environmental Contamination and Toxicology. Springer, New York, NY, p 91–124
Harrison RM, Hester RE (2017) Environmental impacts of road vehicles: past, present and future. Royal Society of Chemistry, London, UK, p 2017
Hwang HM, Fiala MJ, Park D, Wade TL (2016) Review of pollutants in urban road dust and stormwater runoff: part 1. Heavy metals released from vehicles. Int J Urban Sci 20(3):334–360
Hwang HM, Fiala MJ, Park D, Wade TL (2018) Review of pollutants in urban road dust and stormwater runoff: part II. Organic contaminants from vehicles and road management. Int J Urban Sci:1–19. https://doi.org/10.1080/12265934.2018.1538811
Kennedy P, Gadd J, Moncrieff I (2002) Emission factors for contaminants released by motor vehicles in New Zealand. Prepared for the New Zealand Ministry of Transport and Infrastructure Auckland
Kukutschová J, Moravec P, Tomásek V, Matejka V, Smolík J, Schwarz J et al (2011) On airborne nano/microsized wear particles released from low-metallic automotive brakes. Environ Pollut 159(4):998e1006 Elsevier
Kukutschová J, Filip P (2018) Review of brake wear emissions: a review of brake emission measurement studies: identification of gaps and future needs. Non-Exhaust Emissions, p 123–146
Kuo CY, Wang JY, Liu WT, Lin PY, Tsai CT, Cheng MT (2012) Evaluation of the vehicle contributions of metals to indoor environments. Journal of Exposure Science and Environmental Epidemiology 22(5):489–495
Lee PW, Filip P (2013) Friction and wear of Cu-free and Sb-free environmental friendly automotive brake materials. Wear 302(1–2):1404–1413

Needleman HL, Schell A, Bellinger D, Leviton A, Allred EN (1990) The long-term effects of exposure to low doses of lead in childhood: an 11-year follow-up report. N Engl J Med 322(2):83–88

Padoan E, Ajmone-Marsan F, Querol X, Amato F (2017) An empirical model to predict road dust emissions based on pavement and traffic characteristics. Environ Pollut 237:713–720

Padoan E, Amato F (2018) Vehicle non-exhaust emissions: impact on air quality. In: Non-Exhaust Emissions, pp 21–65. https://doi.org/10.1016/B978-0-12-811770-5.00002-9

Panko J, Kreider M, Unice K (2018) Review of tire wear emissions: a review of tire emission measurement studies: identification of gaps and future needs. Non-Exhaust Emissions, p 147–160

Penkała M, Ogrodnik P, Rogula-Kozłowska W (2018) Particulate matter from the road surface abrasion as a problem of non-exhaust emission control. Environments 5(1):9

Root RA (2000) Lead loading of urban streets by motor vehicle wheel weights. Environ Health Perspect 108(10): 937–940

Sanders PG, Xu N, Dalka TM, Maricq MM (2003) Airborne brake wear debris: size distribution, composition, and a comparison of dynamometer and vehicle tests. Environmental Science Technology 37:4060–4069

Steiner M, Boller M, Schulz T, Pronk W (2007) Modelling heavy metal fluxes from traffic into the environment. J Environ Monit 9(8):847–854

Thorpe A, Harrison RM (2008) Sources and properties of non-exhaust particulate matter from road traffic: a review. Sci Total Environ 400(1):270–282

TxDOT. 2018. Houston District Statistics. Retrieved from https://www.txdot.gov/inside-txdot/division/finance/discos.html?dist=HOU

US EPA. (N.D.) IRIS. Integrated risk information system (IRIS). United States Environmental Protection Agency https://www.epa.gov/iris (accessed 2/2019)

US DOT. (2018). Bureau of Transportation Statistics. National Transportation Statistics, United States Department of Transportation. Retrieved from https://www.bts.gov/sites/bts.dot.gov/files/docs/browse-statistical-products-and-data/national-transportation-statistics/223001/ntsentire2018q4.pdf

Wang Q, Zhao HH, Chen JW, Gu KD, Zhang YZ, Zhu YX, Zhou YK, Ye LX (2009) Adverse health effects of lead exposure on children and exploration to internal lead indicator. Sci Total Environ 407(23):5986–5992

Washington State Department of Ecology. (N.D.) Lead wheel weights are banned. https://fortress.wa.gov/ecy/publications/SummaryPages/1207029.html. Accessed 19 Mar 2019