Cost Benefit Analysis of Externality Factors for Battery-Electric Transit Buses

Sustainability Economics (IDM 7090 G05)

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Executive Summary

The potential movement away from conventional diesel-powered buses and toward battery-electric buses for transit operations has become a topical subject across North America and around the world. The motivations for this transition have been primarily based on economic and environmental rationales. While the potential economic viability of electric transit buses has been the subject of considerable discussion, externality factors have not been well addressed in any systematic or fully transparent manner. The purpose of this work thus has been to examine twelve relevant environmental, social and infrastructure externalities, and to quantify their respective monetized values relative to the transition from diesel toward electrified transit buses, as would be included as part of any comprehensive cost-benefit analysis (CBA). By its nature, this work is preliminary and is not fully exhaustive, involving in each case a review of available literature. The intent has been more to gain a better understanding of the relative current importance of different factors, as well as their overall contribution. A variety of simplifying assumptions are employed, with results also based on the application of electric buses specifically in Winnipeg, Manitoba.

A summary of the results for all twelve externality factors is presented in **Table 1**, in order of contribution. All data are provided on a present value, per-bus basis. Nine factors provide a positive contribution (**black**); three provide a negative contribution (**red**). Results show three factors are clearly most important in terms of magnitude. The largest contribution is from greenhouse gas (GHG) emission reductions, and is no surprise. Using Manitoba's clean-grid, electric buses provide fully a 98% reduction of GHG emissions compared to diesel. A second major contribution is provided through avoiding the price volatility associated with diesel fuel. This may appear at first glance surprising, but makes complete sense. Diesel has become an increasingly expensive fuel with unpredictable prices on a week-to-week basis, whereas there are no such price-surprises with electricity. The uncertainty associated with diesel prices already imposes a significant cost, albeit one not likely fully recognized. A third major contribution is significant noise reduction by electric buses, emphasizing their quiet nature. Perhaps surprising, this ranks much higher than for reductions of air pollutants.

Table 1: Summary of Monetized Values for All Externality Factors Investigated			
Component Issue	Present Value of Impact on Electric Bus		
	Lower Travel	Average Travel	Higher Travel
1. Greenhouse Gas Emissions	$21,250	$30,360	$42,500
2. Diesel Fuel Price Volatility	$10,660	$15,230	$21,320
3. Noise Impacts	$11,990	$17,280	$23,980
4. Rare Battery Mineral Scarcity	($8,300)	($8,300)	($8,300)
5. Weight-Induced Infrastructure Damage	($5,520)	($7,880)	($11,040)
6. Photochemical Smog Precursor Emissions	$1,970	$2,810	$3,940
7. Particulate Matter Emissions	$480	$690	$960
8. Carbon Monoxide Emissions	$160	$220	$320
9. Air Toxics Emissions	$130	$180	$260
10. Used Diesel Engine Lubrication Oil Disposal	$110	$150	$220
11. Acid Precipitation Emissions	Negligible	Negligible	Negligible
12. Used Battery Final Disposal	Negligible	Negligible	Negligible
Overall Net Present Value Total	$32,930	$50,740	$74,160

The net overall present value of combining contributions for all issues is positive for electric buses, compared to conventional diesel buses. The net value depends on the assumed annual travel, ranging from about $33,000 overall for lower assumed annual travel to about $74,000 overall for higher assumed annual travel. Indeed, the net overall result depends more strongly on annual travel than any individual component issue on its own. This emphasizes the need, in terms of economic viability, to have electric buses travel as much as is realistically possible.

Several additional findings emerge from the analyses, summarized as follows:

- Although noise reduction provides an important positive contribution, electric buses can be considered as too quiet, emphasizing the need for acoustic alert systems.
- Nitrogen oxides represent the most important of the air pollutant-related component issues, at least in terms of Winnipeg and Manitoba. Levels for this pollutant in Manitoba, particularly within Winnipeg, have been noted as relatively high nationally.
- Two component issues, used lubrication oil disposal and acid precipitation emissions, had been important in the past but have now been largely addressed, through stewardship programs and ultra-low sulphur diesel (ULSD) fuel respectively.
- Two cautionary issues are identified needing closer scrutiny in the future: corporate social responsibility associated with rare battery minerals extraction; and processes for recovery and recycling of battery components in terms of final battery disposal.

1. Introduction

For the past several years, the most advanced battery-electric transit bus project in Canada has been underway in Winnipeg, Manitoba. This demonstration also has been one of the most advanced such projects in the world so far, incorporating four second-generation electric buses, including on-route rapid charging, operating in for-fare service on a regular route with Winnipeg Transit over multiple years (see photograph below presented as **Figure 1**). Background on this project, as well as the development of the earlier prototype electric bus, are described further in a recent report by EVTEC (2017) and by the City of Winnipeg (ND). There are obvious economic benefits and costs that can be calculated for the operation of electric buses compared to conventional diesel buses, these based on the differing individual circumstances of transit authorities across Canada, and elsewhere. The generic benefits of electric buses include much-lower annual fuel cost and lower annual maintenance cost, although balanced against a still typically higher initial purchase cost. The movement toward electrified transit has become topical in both the popular media and literature (e.g., Li 2014 or Doluweera et al. 2017).

Less obvious in the evaluation of electric buses are the effects relative to environmental, social and infrastructure externalities, and their monetized values, whether positive or negative, associated with moving toward electric buses. During 2017, MBA students in the I.H. Asper School of Business at the University of Manitoba, studying Sustainability Economics (Course IDM 7090 G05), examined a wide range of externality factors. This report provides a compendium of the work undertaken by individual students and more, covering twelve distinct identified component issues. Although each one is covered separately, there are some obvious relationships between certain issues, as discussed later. Lastly, this report is presented through the auspices of the Canadian Urban Transit Research and Innovation Consortium (CUTRIC). The authors thank CUTRIC for their assistance and cooperation in this regard.

Figure 1. Photograph of second-generation battery-electric bus on-route in service operation with Winnipeg Transit on Graham Transit Mall, Winnipeg, Manitoba, (Feb 2017 by R. Parsons)

2. Methods

Twelve distinct component issues (each bolded) are included as part of this work in the following areas, presented here in no priority order:

- **Greenhouse gas emissions** (GHG). These primarily involve carbon dioxide but also other such identified constituents as well (see page 4);
- Air pollutant emissions. These involve five so-called "criteria air contaminants" (EPA 2015) including: **carbon monoxide** (CO) (see page 28), nitrogen oxides (NOx) and volatile organic compounds (VOC) grouped together as **photochemical smog precursors** (see page 22); sulphur oxides (SOx), primarily sulphur dioxide, considered as **acid precipitation** (see page 34); and **particulate matte**r (PM), primarily focussing on PM2.5, i.e., particulates less than 2.5 μm (microns) effective diameter (see page 25). **Air toxic** constituents are also considered (see page 29).
- **Used lubrication oil** disposal (see page 32).
- **Noise** (and odour) implications (see page 13).
- **Rare mineral scarcity** related to batteries and other electric components (see page 16).
- **Final battery disposal** (see page 36).
- Weight-caused **infrastructure damage** due to weight differences (see page 19).
- **Price volatility** differences for diesel fuel versus electricity (see page 7).

In all cases, the nature of the issue is explained, as well as whether the impact of moving to electric buses is positive or negative. In terms of estimating associated benefits or costs, a review of existing and relevant literature has been undertaken for each component issue, with all information presented being from public-domain sources. For quantification, a series of major assumptions are employed, as discussed in the next section, to directly compare an electric bus to a diesel bus. All derived benefit and cost results from analyses of all component issues are provided consistently on a present value, per-bus basis. Each issue is presented separately in order of relative monetary significance. Although described separately, there are obvious relationships between some issues, and these are noted. In some cases, perhaps surprisingly, the component issues are found to have little effect one way or another. In most such cases the findings reflect that mitigating measures already have been undertaken. One such example is the implementation of ultra-low sulphur diesel (ULSD) fuel since 2006, which has already significantly reduced sulphur-related emissions from diesel fuel.

This work obviously is preliminary in nature and is not exhaustive, involving in each case a review of available public-domain literature. The results are based on the situation in Winnipeg, Manitoba, and may not be directly applicable to other locations. Simplifying approximations and assumptions are used throughout that, although reasonable, would need to verified or corrected as required for other locations. The primarily objectives of this work, as such, are firstly to assess the overall monetized effect of externalities on viability and desirability of the transition to electric buses, and secondly to provide a relative assessment of which factors may be more important. As more in-depth analysis is undertaken, improved results can be obtained.

3. Major Basic Assumptions

Financial-related assumptions are dominated by estimating an appropriate value for the present value interest factor annuity (PVIFA), which is used to convert between annual future operating costs and present value costs. This parameter in turn requires a stipulated project life and cost of money. As outlined by Laver et al. (2007) the anticipated lifespan for a standard, 40-foot transit bus is about 12 years. As outlined by Deloitte (2014), the cost of money employed for major recent transit-related projects by the City of Winnipeg has been 4.35%. This is an appropriate surrogate to use for transit buses and their associated infrastructure. The resulting PVIFA value is thus approximately 9.20. Where required, a consistent currency conversion rate of 0.80 U.S. dollar per Canadian dollar has been used.

Energy consumption and operational assumptions for electric versus diesel buses are as outlined by EVTEC (2017). In this case, a diesel bus consumes approximately 62 Litres per 100 km of blended fuel, which corresponds to the current average fuel consumption by diesel buses within Winnipeg Transit. In terms of fuel composition, all diesel fuel within Manitoba is mandated to contain a minimum of 2% biodiesel, which is assumed. A uniform density value of 0.83 kg per Litre is also assumed for diesel and diesel fuel blends. An electric bus consumes approximately 160 kWh per 100 km of electricity on average for year-round operation. A small amount of diesel fuel blend is also assumed for auxiliary heating during the coldest weather conditions to preserve battery state-of-charge.

The extent of annual travel is an obvious important variable, given that higher travel displaces more diesel fuel. Three assumed values are included, which are also as outlined in EVTEC (2017): (i) **Lower annual travel**, corresponding to 35,000 km per year; (ii) **Average annual travel**, corresponding to 50,000 km per year and also representing the current average travel per bus by Winnipeg Transit; and (iii) **Higher annual travel**, corresponding to 70,000 km per year. For direct comparison of electric versus diesel buses, available public-domain data has been accessed from so-called "Altoona" testing results for bus models. This testing, involving accelerated performance and durability evaluation, is undertaken by a third-party entity associated with Pennsylvania State University, and is required for any bus model to be eligible for funding from the U.S. Federal Transit Administration. Altoona test results are available for two directly comparable standard models of New Flyer's advanced Xcelsior bus platform:

- **Electric version** (XE40), as outlined in Pennsylvania State University (2015); and
- **Diesel version** (XD40), as outlined in Pennsylvania State University (2012)

Two important statistics in the evaluation of a number of the component issues are firstly bus average speed, and secondly bus average passenger loading. In terms of speed, a uniform value of approximately 20 km per hour is assumed. This value is based on data from APTA (2015) showing overall average speed across the U.S. for all transit buses to be 12.5 miles per hour (see their Table 6), which is exactly the same as 20 km per hour. In terms of passenger loading, overall data from APTA (2015) suggests that when considering revenue miles, average loading is approximately 10 passengers. This calculation is provided as follows:

Average Loading = 19,400 million passenger miles (their Table 5) ÷ 1,935.3 million vehicle miles (their Table 6) = 10.02 passengers per vehicle (transit bus)

Passenger capacity for the electric bus (XE40 as tested, see above) is 76 in total (i.e., 39 seats plus 37 standees). Passenger capacity for the diesel bus (XD40 as tested, see above) is 81 in total (i.e., 35 seats plus 45 standees). For loading capacity purposes, the total capacity of the electric bus version has been used for reference, such that an average of 10 passengers translates to 13.2% of overall bus capacity. Although the diesel bus version does have roughly 7% larger overall capacity, the electric bus version has 11% more seats, which provides a higher quality ride for passengers.

For many of the air pollutant related impacts considered, consistent emission factor data have been available from the U.S. (Cai et al. 2013). Emission factors for modern diesel transit buses are provided (on expected model-year by model-year basis). Their data are presented on a g per mile basis, with data translated to a g per km basis prior to use. The health impacts of diesel exhaust are important across a variety of air pollutants considered in this analysis. Health Canada (2016) recently completed a human health risk assessment for diesel exhaust, which provides extensive background as well as healthcare-associated adverse outcome and cost information. This authoritative source has been used across a number of the issue-categories considered, including photochemical smog precursors (see page 22), particulate matter (see page 25), and air toxics (see page 29).

4. Greenhouse Gas Emissions

The most significant external factor affecting electric buses is their reduction of greenhouse gas (GHG) emissions. This is positive for the transition from diesel to electricity. Although this aspect has been recognized for some time, the announcement by the Government of Canada (2016) to implement a minimum price-on-carbon, means that a uniform direct cost will shortly come into effect. As outlined in ECCC (2017), the "floor price" for carbon starts at a level of $10 per tonne beginning January 2018, rising $10 per tonne each year, and reaching $50 per tonne in 2022, all presented on a nominal basis. No specific policy has been yet outlined for subsequent years, but it is reasonable to assume at minimum the price level will remain at least at $50 per tonne. Based on this schedule of costs combined with the assumed cost of money, it is straightforward to calculate the present value for a unit rate of emissions over time, and then back-determine a future year average carbon price, which translates to $40 per tonne.

The recent work by EVTEC (2017) includes estimates of GHG reductions for electric buses operating in Manitoba (see their Appendix B). Importantly, they note that the extent of emission reductions depends significantly on both the annual travel of the bus, and the method by which emissions are calculated, with three main approaches noted. The first approach considers only emissions from combustion at the vehicle itself, often termed "tank-to-wheels" (TTW), or "pump-to-wheels" (PTW). The second approach involves full-cycle emissions, including upstream emissions from the exploration, extraction and refining of fuels and inputs, often termed "well-to-wheels" (WTW). Both have been used in academic work, however, neither actually reflects how emissions are tabulated for individual jurisdictions as part of Canada's annually updated National Inventory Report (NIR). This third approach, based on NIR, considers only the emissions that occur within the jurisdiction. In terms of buses this effectively translates to diesel fuel and related inputs being considered on a combustion-only basis, and electricity being considered on a full-cycle basis, given it is typically generated entirely within the jurisdiction.

EVTEC (2017) presents emission reduction estimates based on all three approaches. It is obviously most advantageous to consider electric buses on a full-cycle basis. Based on this approach, for example, annual reductions for a single high-travel bus exceed 160 tonnes. Only the third (NIR approach), however, is considered here given how the price-on-carbon is applied. Diesel bus emissions consider the use of a 2% blend of biodiesel in diesel fuel, as mandated in Manitoba, and additional diesel emission fluid (DEF) for operation of Tier 4 post-exhaust treatment. The resulting emissions translate to 165 kg per 100 km, based on NIR. Electric bus emissions involve use of electricity from Manitoba's grid mix and a small amount of diesel blend for auxiliary heating. This translates to 3 kg per 100 km, based on NIR. The resulting net reduction is thus 162 kg per 100 km. This represents a reduction of more than 98% compared to diesel, which is highly significant. Importantly, the effect of using a small amount of diesel for auxiliary heating is trivial, and makes sense given application in efficient catalytic heaters.

The annual savings in GHG emission costs per electric bus in a future average year can be calculated based on variable annual travel, as follows:

- **Lower annual travel**: 35,000 km × 162 kg/100 km × $40/tonne = $2,268 per year
- **Average annual travel**: 50,000 km × 162 kg/100 km × $40/tonne = $3,240 per year
- **Higher annual travel**: 70,000 km × 162 kg/100 km × $40/tonne = $4,536 per year

The resulting present value savings over the 12-year life of a typical bus translate to the following:

- **Lower annual travel**: $2,268 per year × 9.20 PVIFA = $20,870 present value
- **Average annual travel**: $3,240 per year × 9.20 PVIFA = $29,810 present value
- **Higher annual travel**: $4,536 per year × 9.20 PVIFA = $41,730 present value

Manitoba's clean electricity grid means that significant savings are achieved per electric bus based on the currently proposed Federal price-on-carbon. Two obvious subsidiary questions arise on the subject of GHG emission reductions. The first question is, what about savings from emission reductions for electric buses in other jurisdictions, especially where the grid-mix is not so clean as in Manitoba? This is addressed directly in the text box on the next page.

The second question is, what about the impact of emission reductions for an electric bus compared to a modal-shift, i.e., simply getting people out of passenger cars and into public transit? This is addressed directly in the text box at the top of page 7. Evaluating this latter question also requires understanding the comparative emission levels for light duty vehicles, which overwhelmingly use gasoline as fuel, versus diesel buses.

Within Manitoba, average fuel consumption for light duty vehicles of all types is relatively high, around 15 Liters per 100 km. This reflects not just cold weather operations, but a significant transition underway for some time away from passenger cars and toward larger units (i.e., pick-up trucks, vans and SUVs), all of which show higher fuel consumption than cars. If just a typical passenger car is considered, the suitable fuel consumption is lower, around 10 Litres per 100 km. Emissions per vehicle in these cases, including Manitoba's current 8.5% ethanol mandate, translate to approximately 33 kg per 100 km for an average light duty vehicle and 22 kg per 100 km for a typical car. Compared to these, the emissions for a diesel bus are roughly equivalent to 5 vehicles or 7.5 cars respectively per 100 km. At the same time, buses travel further in a given year, on average 50,000 km annually as noted earlier, compared to around 16,000 km annually for a light duty vehicle. Assuming these average annual travel values, a diesel bus generates approximately 82.5 tonnes annually, compared to 5.3 tonnes annually for a light duty vehicle (i.e., single bus equivalent to roughly 16 vehicles) or to 3.5 tonnes annually for a typical car (i.e., single bus equivalent to roughly 24 cars). As such, in approximate terms, the electrification of a single transit bus is equivalent to around 20 light duty vehicles, which is significant.

Electric Buses in Other Jurisdictions?
The ability to achieve emission reductions via electric buses depends on the nature of the grid-mix within the given jurisdiction. Manitoba has highly clean electricity, with average GHG emissions of just 4 g per kWh, but the nature of the grid-mix varies dramatically across Canada. Relatively recent information is available for 2014 (Environment Canada 2016). The average grid-mix electricity emissions for the country as a whole at that time represent approximately 160 g per kWh. Six jurisdictions are well below this level and in all cases, just like Manitoba, electric buses readily can make sense in terms of emission reductions. From west to east, these are: British Columbia, Yukon, Manitoba, Ontario, Quebec, and Newfoundland and Labrador. Three jurisdictions have intermediate electricity emissions in the range of 200 to 400 g per kWh, namely New Brunswick, Prince Edward Island, and Northwest Territories. Lastly, four jurisdictions have higher electricity emissions, greater than 700 g per kWh, namely Nova Scotia, Alberta, Saskatchewan, and Nunavut.
Nunavut likely has the highest emissions, but represents a special case, firstly given electricity generation there is dominated by diesel and secondly, given its small size, their generation has simply been aggregated with the Northwest Territories and is not separately reported. Of the others, Alberta and Saskatchewan share the next highest emission level, at 820 g per kWh. Based on this, emissions for an electric bus, again including a small amount of diesel blend for auxiliary heating, translate to 134 kg per 100 km, based on calculation methods and assumptions used by EVTEC (2017). This means there is still a reduction compared to a diesel bus of 31 kg per 100 km. Even in these cases, electric buses achieve emission reductions of close to 20%. As such, it can be clearly stated that electric buses can achieve emission reductions virtually everywhere across Canada, irrespective of grid-mix. What varies is the extent of reduction. At the same time, it is important to note that for jurisdictions with higher grid-mix GHGs, other alternative solutions may achieve better emission reduction results.

Electric Buses versus Modal Shift?
An important aspect of electric buses is that emission reductions achieved through transitioning from diesel to electricity as motive energy carrier are entirely separate from and in addition to the more-traditionally understood emission improvements gained from buses, which are namely through modal-shift. This involves transitioning people away from using individual vehicles toward using buses (or other public transportation conveyances). But how do they compare? Quantifying and comparing these reductions requires determining the corresponding emissions on the basis of per passenger-km travel. Such values are more complex to estimate, involving a larger number of assumptions, as well as ongoing variability that occurs in passenger loadings, and, as such, they are less commonly presented in the literature. It is obvious that the best possible reduction to be achieved is by targeting single occupancy vehicles (SOV). Importantly, Hodges (2010), on behalf of the U.S. Federal Transit Administration, indicated that in 2008 the average carbon dioxide emissions from a SOV in the U.S. averaged 0.964 lb per mile. This translates to about 274 g per km (same as passenger-km for SOV). Based on the fifty largest transit fleets in the U.S., bus-based carbon dioxide emissions in the same year were an average of 0.643 lb per passenger-mile, with data weighted on a passenger-mile travel basis. This translates to about 183 g per passenger-km. As such, each passenger moving from a SOV to a bus reduces emissions in the U.S. by approximately one-third. Additional, more recent, emission factors have been presented by the EPA (2015). They indicate emissions translate to approximately 228 g per km for a typical car and 303 g per km for a typical light truck (same as passenger-km for SOV). These data straddle results of Hodges (2010). At the same time, they suggest emissions for a bus to be about 34 g per passenger-km, although not distinguishing between transit versus inter-city coach buses. The former have much higher fuel consumption compared to the latter, so the value is less relevant.
Within Manitoba, average blended-gasoline fuel consumption for light duty vehicles has remained high, around 15 Litres per 100 km, as described earlier. Emissions per vehicle in this case translate, including Manitoba's current 8.5% ethanol mandate, to approximately 330 g per km (same as passenger-km for SOV). This is somewhat higher than Hodges (2010), but not unexpected, given the trend here away from cars to pick-ups, vans and SUVs. Using Manitoba-based bus emissions data as calculated by EVTEC (2017) and assuming a reasonable approximate average loading of 10 passengers for an average travel bus, means that diesel bus emissions are approximately 165 g per passenger-km. This is relatively consistent with Hodges (2010). For the electric bus, emissions are only approximately 3 g per passenger-km, less than 1% of that for an average SOV. Based on these results, a modal-shift of passengers from SOV to diesel buses reduces emissions by approximately one-half, while the further reduction by moving the bus from diesel to electricity is almost the same. The emission reductions achieved in different jurisdictions obviously may vary somewhat, but this analysis clearly shows the relative overall importance of transitioning to electric buses. Further, the reduction achieved through electrification of bus operation is directly under the control of the transit authority and is much more certain in terms of achievability, i.e., not subject to uncertain market responses.

5. Diesel Fuel Price Volatility

Price volatility is a complex but subtle component issue that is positive for the transition from diesel to electricity as the motive energy carrier for buses. Analysis shows, perhaps surprisingly, this to be important in terms of its monetized value. Conventional business case analyses typically involve the use of a consistent diesel fuel price over the vehicle lifespan, even potentially with prices escalating, but in all cases in a set and predictable manner. Volatility analysis, on the other hand, characterizes the unpredictable nature of market fuel price movements on a week-to-week basis. As described in more detail, on a theoretical basis, this is the second most important monetized component issue, however, on a more practical basis, costs would be reduced somewhat, although still being the third highest monetized value.

Electricity prices have been well known across North America to be both relatively more stable and relatively more predictable when compared to the prices of fossil fuels, including diesel. This was not always the case, however, in that during the late 1990s fossil fuel prices, diesel prices in particular, were relatively stable, with little fluctuation at the retail pump price on a week-to-week basis. This situation began to change dramatically around 2000, as illustrated by average fuel price data for the U.S. in **Figure 2** (average prices presented on gasoline gallon equivalent basis using data from AFDC 2017). Gasoline and diesel prices both began to steadily rise, more than doubling by 2008. Since then, fossil fuel prices have varied significantly, albeit with no long-term direction, either upward or downward. Fossil fuel prices thus today remain highly variable. From a financial perspective, such uncertainty and unpredictability translates ultimately to higher costs.

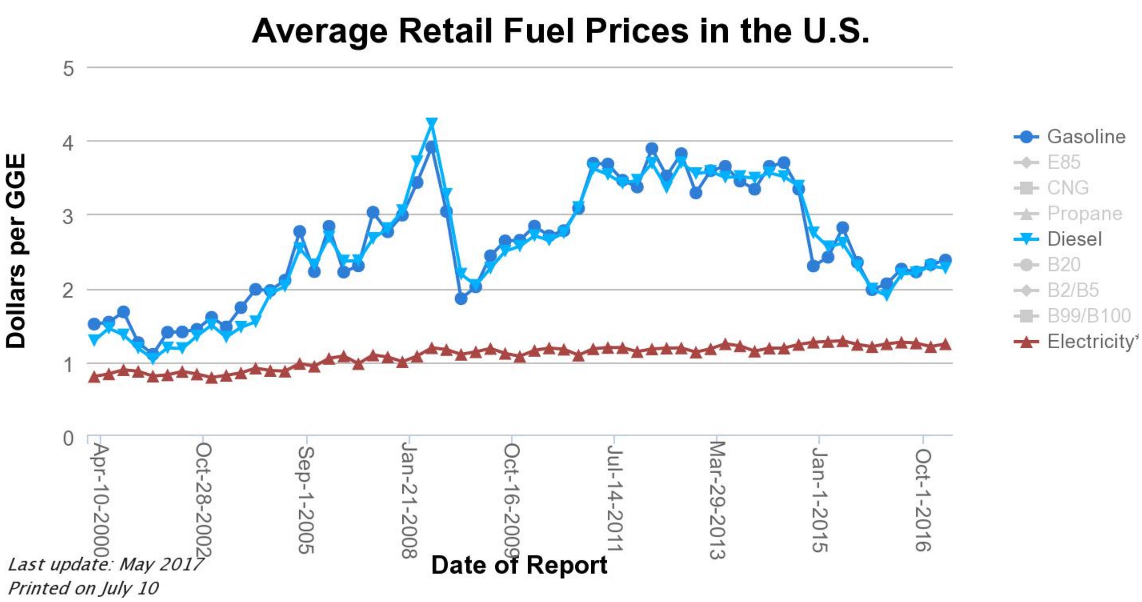

Figure 2. Average Retail Fuel Price Equivalents in U.S. from 2000 through 2016 for diesel, gasoline and electricity on gallon gasoline equivalent (GGE) basis from AFDC (ND)

At the same time, the relative pricing of diesel versus gasoline has also changed significantly. Diesel traditionally was much less expensive than gasoline (on volumetric basis) across North America, however, over the past two decades as described by Parsons and Cottes (unpublished), diesel has progressively become more frequently the most expensive fossil fuel (i.e., majority of time on a volumetric basis and increasing proportion of time on an energy content basis). In sharp contrast to traditional expectations, diesel has been becoming both relatively unpredictable in terms of week-to-week price movements and relatively more expensive compared to gasoline.

Manitoba provides a unique and useful case for comparison given that there is effectively no market uncertainty and thus no inherent risk with regard to unexpected variability of electricity prices for customers here. Manitoba has a single, Crown electrical utility, Manitoba Hydro, which is regulated. A full schedule of rates and chargers is provided and any changes in rates must be approved first by the Public Utilities Board. Although it is true electricity costs can increase, for example through demand charges associated with varying rapid charger loads, the nature of these costs is completely stipulated in advance and they can be managed at the customer's discretion. On the other hand, future diesel prices are simply unknown.

The emerging problematic nature of diesel as a fuel was identified by the transit industry as early as 2008 (APTA 2008). A survey of transit authorities in that year showed the unit price of diesel, as used in diesel buses, had increased by a total of 166% over the previous four-year period (from 2004 to 2008). Over the same period the price of electricity, as used in trolley buses, increased by a total of only 19%. The increase in diesel prices forced almost a doubling of the proportion of agency budgets that had to be devoted to cover energy (i.e., from 6% to 11%). The most common responses included fare increases (48%), increased state and local contributions (43%), delays in operating and capital improvements (42% each), delays or cancellation of service increases (38%), and funding transfers from capital to operating budgets (38%). The realization of concerns with diesel prices, and associated negative impacts on transit operations, also led directly to the development through the Transportation Research Board of a more-formal guidebook on fuel price strategies, including price-hedging, specifically for transit authorities (Friedman and DeCorla-Souza 2011).

Derivatives represent a class of financial instruments whose value depends on the value of an underlying variable (Hull 1997), often the price of an asset or commodity, like diesel fuel. Options are a form of derivative that first came into use during the 1970s, and today are frequently used to address (or hedge) risks. A call option is the appropriate hedging approach to employ for this calculation, given that it is indeed used to protect against rising commodity prices. A call option provides the right but not the obligation to buy the underlying commodity by a certain time (exercise date) for a certain value (strike price). The Black-Sholes equation provides the fundamental basis for the valuation of options, as summarized by Hull (1997). In turn, this equation is based on Brownian motion, the random movement of particles in a fluid. The higher the variation in the price of the underlying commodity, the greater the possibility that a strike price can be viably achieved at more extreme price values. In other words, higher variation in a commodity's price means a higher value for the option derivative. This provides a method to quantify the cost of volatility in diesel prices. As described by Hull (1997), the value of an option derivative using the Black-Sholes equation depends on a series of key defined variables, as follows:

- Risk-free rate, assumed as the normal discount rate of 4.35%;
- Timeframe for the exercise date, assumed simply as one-year in all cases;
- Standard price of the underlying commodity (i.e., the price of diesel), discussed later;
- Strike price, or the ratio of strike price to standard price, which corresponds to the maximum tolerable price increase on an annual basis, as discussed above; and

- Estimated annual volatility of the underlying commodity price, which in this case is derived from the extent of observed weekly changes in retail diesel prices.

Being able to define the last of these variables requires adequate relevant diesel pricing data. As described by Parsons and Cottes (unpublished), Natural Resources Canada (NRCan) had until recently provided explicit data on average weekly diesel (and other fuel) retail prices for individual cities, including Winnipeg, as well as the average for the country as a whole. A further suggestion from Hull (1997) regarding evaluation is that the data used for price volatility determination should be at least as extensive as the period of time to be considered into the future, hence 12 years. Twelve years of data are thus used, as had been obtained earlier from the Fuel Focus website of NRCan (2015). These data are specifically for diesel in Winnipeg and cover 2003 through 2015.

Based on 626 total data points, the standard deviation for the natural logarithm of the ratio of average price in each week to average price the previous week can be determined for all sequential data points. This statistic is found to be 0.0175. This data is illustrated visually in **Figure 3**, showing the ongoing extent of variations. Dividing this value by the square root of the time interval for the data points in units of years (i.e. 1 ÷ 52 = 0.0192, with square root value of 0.1388) provides an estimate for volatility of 0.126, or 12.6% annually. The standard error of the estimate in this case is calculated as 0.04%.

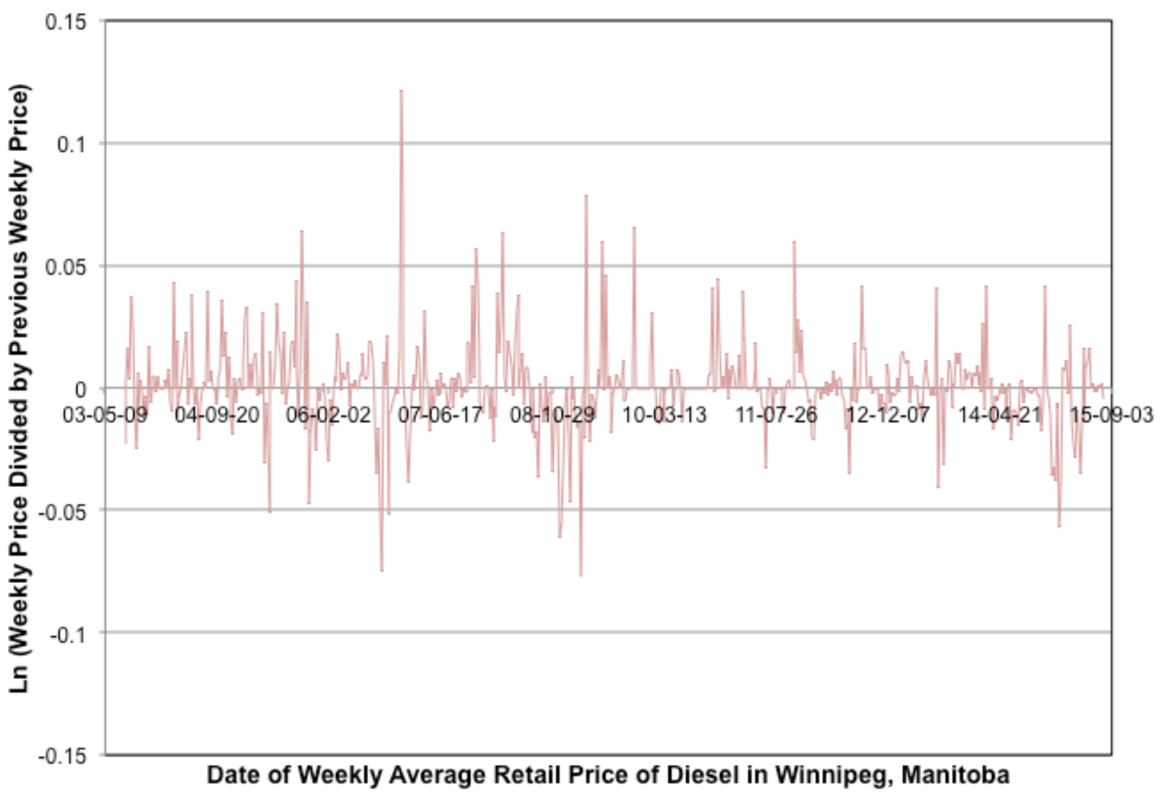

Figure 3. Natural logarithm of average weekly Winnipeg retail diesel fuel price divided by the average price for the previous week for 2003 through 2015

The calculation also depends on the specific standard price used for diesel. For this purpose, the selected value is the mean retail diesel price for Winnipeg covering 2008 through 2015, which, as described earlier by Parsons and Cottes (unpublished), had been a "variable price period" without any significant overall price trend either upward or downward. This value is \$1.12 ± \$0.16 per Litre (n = 395), and is also understood to be reasonably proximal in terms of applicability for transit too.

The approach in this case is based on purchase of a single call option derivative per year on a per-Litre diesel fuel basis. The strike price represents the maximum permissible price that can be tolerated internally by the transit authority, with any fuel price increases above that point covered by the call option. To fully assess the impact of volatility, it is at least theoretically valid as a starting point to use the standard price itself as the strike price, i.e., ratio of strike price to standard price = 1.00. As already noted, conventional business case evaluations typically assume a consistent diesel price, or possibly an escalating price, but in all cases predictable week-to-week and not taking account of any unexpected price increases.

The Black-Scholes equation, as employed for this case, is outlined by Hull (1997, see their page 241), with derivation and calculations presented in detail, as follows, for the case where the strike price equals the standard price:

Equation parameters d_1 and d_2:
$d_1 = [\ \ln(1/1.00) + (0.0435 - (0.126)^2/2) \times (1)\ ] \div [\ 0.126 \times (1)\ ] = 0.408$
$d_2 = d_1 - 0.126 \times \mathrm{sqrt}(1) = 0.282$

Values of normal cumulative distribution function (i.e., for mean = 0, SD = 1):
$N(d_1) = 0.6584$
$N(d_2) = 0.6110$ (i.e., probability that option will be exercised in a risk-neutral world)

Calculation of price for European call option:
$c = \exp(-0.0435 \times (1)) \times [\ \$1.12/L \times 0.6584 \times \exp(0.0435 \times (1)) - \$1.12/L \times 0.6110\ ]$
$c = \$0.0822$ per Litre

Using these above assumptions, the annual costs, based on fuel consumption for the three annual travel options, and resulting present values can be calculated, summarized as follows:

- **Lower annual travel**: 21,700 Litres per year × \$0.0822 per Litre = \$1,784 per year × 9.20 PVIFA = \$16,410 present value
- **Average annual travel**: 31,000 Litres per year × \$0.0822 per Litre = \$2,548 per year × 9.20 PVIFA = \$23,440 present value
- **Higher annual travel**: 43,400 Litres per year × \$0.0822 per Litre = \$3,568 per year × 9.20 PVIFA = \$32,820 present value

Assuming the strike price to be the same as the standard price for the purchase of call options clearly shows price volatility to be the second most important issue on a monetized basis, at least theoretically. Given this situation, diesel price volatility has been ranked as the second highest issue. However, it is unlikely such an approach as employed above would be used in practice due to excessively high costs.

Including the acquisition of the call option too means that the price of diesel would need to be above $1.20 per Litre, or at a price roughly 7.3% above the standard price, in order for all costs to be recovered. Importantly, the cost of the call option declines progressively as higher strike price values are considered, with the cost dropping below the point where the call option is in the money for a strike price roughly 5% above the standard price, as illustrated in **Figure 4**. This point then becomes effectively the cost optimum, and is considered in more detail. It also makes sense given that there would always be at least some flexibility in toleration of upward price movements for fuel, for which a 5% increase in prices is entirely reasonable.

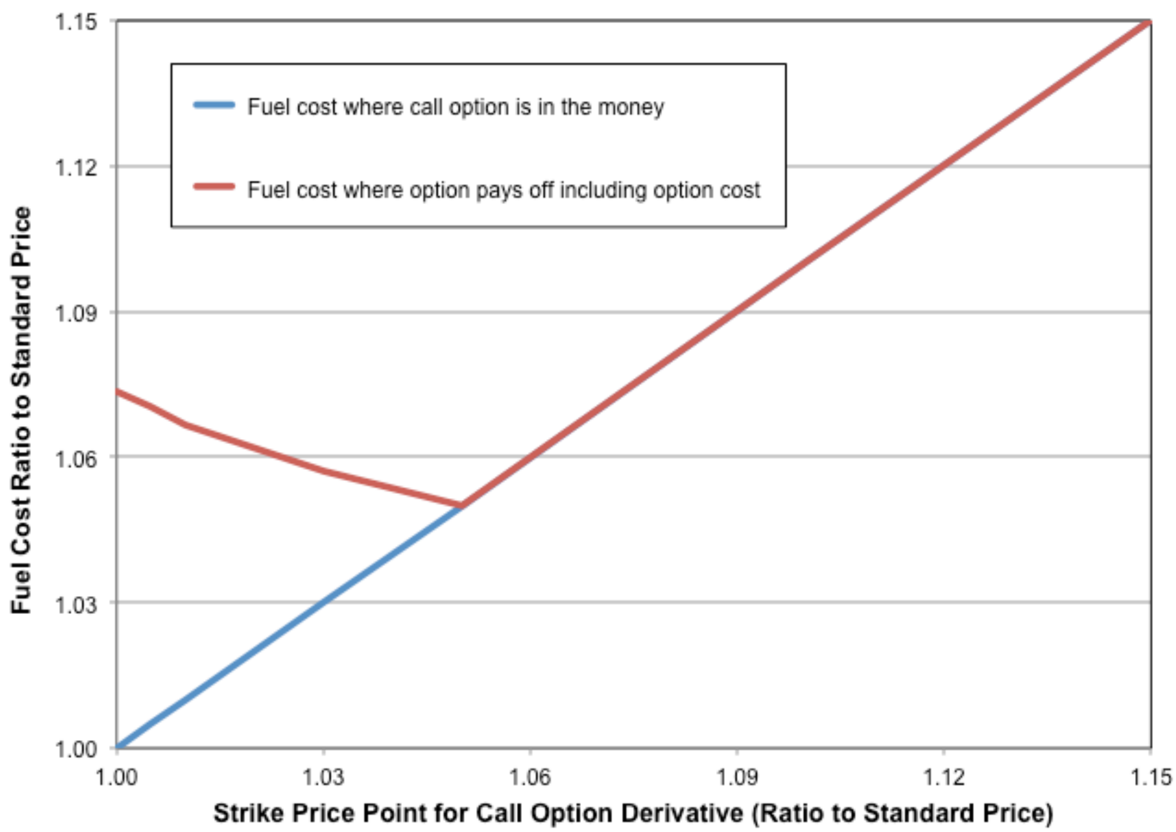

Figure 4. Effective cost of fuel including call option derivative value as a function of the strike-price point (maximum fuel price increase that can be internally tolerated).

Based on using the Black-Scholes equation, the derivation and calculations are presented in detail, as follows, for the case where the strike price is 5% higher than the standard price:

Equation parameters d_1 and d_2:
$d_1 = [\ \ln(1/1.05) + (0.0435 – (0.126)^2/2) \times (1)\ ] \div [\ 0.126 \times (1)\ ] = 0.0286$
$d_2 = d_1 – 0.126 \times \mathrm{sqrt}(1) = -0.0974$

Values of normal cumulative distribution function (i.e., for mean = 0, SD = 1):
$N(d_1) = 0.5114$
$N(d_2) = 0.4612$ (i.e., probability that option will be exercised in a risk-neutral world)

Calculation of price for European call option:
c = exp(-0.0435 × (1)) × [$1.12/L × 0.5114 × exp(0.0435 × (1)) - $1.176/L × 0.4612]
c = $0.0534 per Litre

Using this value, the annual costs, based on fuel consumption for the three annual travel options and resulting present values can be calculated, summarized as follows:

- **Lower annual travel**: 21,700 Litres per year × $0.0534 per Litre = $1,159 per year × 9.20 PVIFA = $10,660 present value
- **Average annual travel**: 31,000 Litres per year × $0.0534 per Litre = $1,655 per year × 9.20 PVIFA = $15,230 present value
- **Higher annual travel**: 43,400 Litres per year × $0.0534 per Litre = $2,318 per year × 9.20 PVIFA = $21,320 present value

The above costs are included in the overall evaluation of externality factors, given they are more practically realistic. Although determined indirectly, based on fundamentals of option valuation, these are costs that already are effectively being incurred in transit operations, although not necessarily explicitly recognized. Essentially the values translate as the costs associated with fuel prices being unpredictably higher than what is expected, with the significant adverse impacts of such occurrences as had been outlined by APTA (2008). The stability and predictability of electricity prices represent a significant and often overlooked advantage in the arena of volatile fossil fuel prices.

6. Noise (and Odour) Impacts

The reduction of noise is a well-identified benefit of electric vehicles in general, and represents an important positive effect for the transition to electric buses, essentially for the reduction of urban traffic noise. The monetized values involved are significant but not overwhelming. At the same time, although diesel vehicles also have been identified to produce potentially objectionable odorants, odour represents a highly individual and subjective matter with little cost data available. As such, quantitative impacts for odour reduction are not included in this report, only those for noise.

Noise effects for electric buses are evaluated in three parts: (i) reduced noise for passengers within the bus, which is more disturbance and health-related; (ii) reduced noise for people in the immediate vicinity of the exterior of the bus, which is also more disturbance and health-related; and (iii) impacts on the values of property along the bus route, which is a commonly considered aspect of traffic noise. For the first two effects, the key parameters are the relative noise levels involved, and the operational exposure. Further, comfortable noise levels have been identified to be in the range of 40 to 60 dB (CERN 2014), such that no monetized costs are considered for noise levels at or below 60 dB.

As part of Altoona-based testing of buses, described earlier, five standardized noise evaluations are included. Results are publicly available for the two relevant New Flyer buses, the XD40 and the XE40 as noted, which are virtually identical in all respects except the first is diesel powered and the second is electric. From an overall perspective the electric bus is generally quieter, with results for the specific tests summarized as follows in **Table 2**, providing relevant differential noise levels (Pennsylvania State University 2012 and 2015):

Table 2. Summary of Altoona-based Noise Tests for Diesel versus Electric Bus Models

Test	Description	Result
1	Buffering passengers inside from 80 dB outside noise level	Electric bus lower noise, averaging 47 dB compared to diesel bus averaging 50 dB, but with both below 60 dB
2	Interior noise while accelerating from 0 to 35 miles per hour	Electric bus lower noise, averaging 68 dB compared to diesel bus averaging 72 dB
3	Exterior noise while accelerating from 35 miles per hour to full throttle	Electric bus lower noise, averaging 66 dB compared to diesel bus averaging 68 dB
4	Exterior noise while accelerating from standstill	Electric bus lower noise, averaging 65 dB compared to diesel bus averaging 69 dB
5	Exterior noise while idling, both with and without accessories active	Electric bus lower noise, averaging 44 dB compared to diesel bus averaging 58 dB, with both below 60 dB so no cost impact

In order to evaluate exposure levels, the operational capacity factor is estimated, i.e., proportion of time bus is operated. These values depend on annual travel with the further assumption of a uniform average 20 km per hour travel speed, as outlined earlier. Results are as follows:

- **Lower annual travel**: 35,000 km ÷ 20 km per hour ÷ 8,760 hours per year = 0.20
- **Average annual travel**: 50,000 km ÷ 20 km per hour ÷ 8.760 hours per year = 0.29
- **Higher annual travel**: 70,000 km ÷ 20 km per hour ÷ 8,760 hours per year = 0.40

Noise levels of buses are dominated by acceleration events, which are outlined in the above data table. Buses, however, are not constantly involved in acceleration mode. As described by Yang (2011), a typical bus is involved in acceleration (as opposed to cruising, deceleration, or idle) only roughly 0.25 of the time. Swedish data on costs of noise, both in terms of disturbance and health effects have been assembled, based on constant noise exposure levels on a per person level (Trafikverket 2015). Intermittency is accounted by the above operational capacity values. Further, an assumed currency conversion of one Swedish kronor equivalent to 0.153 Canadian dollars, and average bus loading of 10 passengers, as outlined earlier, are used.

Interior noise related costs in terms of disturbance and health effects are summarized, based on the difference shown in Test 2 results, as follows:

- **Lower annual travel**: [20,469 SEK (i.e., for 72 dB) – 13,851 SEK (i.e., for 68 dB)] × 0.153 CAD per SEK × 10 per bus average load × 0.20 portion of year × 0.25 proportion in acceleration mode = $506 per year
- **Average annual travel**: [20,469 SEK (i.e., for 72 dB) – 13,851 SEK (i.e., for 68 dB)] × 0.153 CAD per SEK × 10 per bus average load × 0.29 portion of year × 0.25 proportion in acceleration mode = $734 per per year
- **Higher annual travel**: [20,469 SEK (i.e., for 72 dB) – 13,851 SEK (i.e., for 68 dB)] × 0.153 CAD per SEK × 10 per bus average load × 0.40 portion of year × 0.25 proportion in acceleration mode = $1,012 per per year

Exterior noise related costs in terms of disturbance and health effects are summarized as follows, based on the average difference shown in Test 3 and Test 4 results together with the further assumption that on average 10 people external to the bus are impacted separately by each bus (being essentially equivalent to the average bus passenger load):

- **Lower annual travel**: [14,617 SEK (i.e., for 68-69 dB) – 10,610 SEK (i.e., for 65-66 dB)] × 0.153 CAD per SEK × 10 people affected × 0.20 portion of year × 0.25 proportion in acceleration mode = $307 per bus per year
- **Average annual travel**: [14,617 SEK (i.e., for 68-69 dB) – 10,610 SEK (i.e., for 65-66 dB)] × 0.153 CAD per SEK × 10 people affected × 0.29 portion of year × 0.25 proportion in acceleration mode = $444 per bus per year
- **Higher annual travel**: [14,617 SEK (i.e., for 68-69 dB) – 10,610 SEK (i.e., for 65-66 dB)] × 0.153 CAD per SEK × 10 people affected × 0.40 portion of year × 0.25 proportion in acceleration mode = $614 per bus per year

In terms of effects on property values, Weisbrod et al. (2008) included a cost factor per passenger car based on Australian data equivalent to negative impact of about 0.7¢ per km of travel per year. (Note that although Weisbrod et al. properly indicates the data as cents per km, a dollar sign is inadvertently included that may cause an incorrect higher value to be assumed). Further assumptions are that a diesel bus is approximately 4× more noisy than a passenger car, while an electric bus is approximately 2× more noisy than a car, with the net implication that an electric bus is roughly half as noisy as a diesel bus The latter is as indicated from available data discussed earlier. An Australian dollar is assumed roughly equivalent to Canadian. Based on these assumptions, annual property damage levels are estimated as follows:

- **Lower annual travel**: 35,000 km × $0.007 per km × (4 – 2) magnifier = $490 per year
- **Average annual travel**: 50,000 km × $0.007 per km × (4 – 2) magnifier = $700 per year
- **Higher annual travel**: 70,000 km × $0.007 per km × (4 – 2) magnifier = $980 per year

Results in terms of present value per bus are summarized as follows:

- **Lower annual travel**: ($506 + $307 + $490) per year × 9.20 PVIFA = $11,990 present value

- **Average annual travel**: ($734 + $444 + $700) per year × 9.20 PVIFA = $17,280 present value
- **Higher annual travel**: ($1,012 + $614 + $980) per year × 9.20 PVIFA = $23,980 present value

Noise reduction has become increasingly recognized as important within modern urban areas. The values are significant, although not overwhelming. At the same time, especially during idling periods, electric vehicles, including electric buses, can be perceived as too silent, and thus potentially pose a hazard. Indeed, as outlined by Feith (2012), a United Nations working group on Quiet Road Transport Vehicles (QRTV) has been working toward requirements for acoustic vehicle alerting systems (AVAS). These are intended in particular to address concerns of visually impaired individuals whose ability to detect the proximity of these such vehicles may be constrained if audible cues are not available.

Quantified impacts of diesel bus odour on populations could not be found in the literature reviews, as undertaken. Odour is detected by the interaction of odorant compounds with sensory receptors in the human olfactory bulb located in upper nasal passages where inhaled air passes. Complexities associated with odorant compounds and the evaluation of odour perception are outlined by Bratolli et al. (2011). Odour perception of specific odorant chemicals can be highly individual and subjective. It has been known for some time that diesel engines can produce objectionable odorant compounds (Cernasky 1983), especially due to the exhaust. However, while initial concerns were characterized in terms of nuisance pollutant characteristics, the more pernicious and damaging nature of diesel exhaust has since come to light. This includes the effects of the criteria air pollutants and air toxics that are present in the exhaust. These aspects of diesel exhaust are discussed in more detail in several later sections.

7. Rare Battery Mineral Scarcity

Concerns with rare mineral scarcity represent an important and negative implication for the transition to electric transit buses. The monetized impacts in this analysis are associated with further potential mineral price escalations, based on recent trends. The overall values involved are significant, although not overwhelming, with the two most important minerals so far being cobalt and lithium.

This is an emerging cautionary area for all battery manufacturers, as well as electric vehicle manufacturers and integrators, to be aware. Exchanging high dependency on one commodity, i.e., diesel petroleum, for high dependency on another commodity, i.e., rare battery minerals, is not intuitively sustainable. A further cautionary aspect of rare minerals is regarding corporate social responsibility (CSR). The extraction of these necessary minerals is undertaken significantly in developing countries under sometimes potentially exploitative conditions.

Although lithium ion batteries are often thought of, at least popularly, as “one thing,” they instead represent a cluster of products with related but varying battery chemistries. As outlined by the

Boston Consulting Group (2010), there are already six major commercialized lithium ion battery chemistries, with others under development (Nitta et al. 2015):

- Lithium-cobalt-oxide (LCO);
- Lithium-nickel-cobalt-aluminum (NCA);
- Lithium-nickel-manganese-cobalt (NMC);
- Lithium-manganese-oxide or spinel (LMO);
- Lithium-titanate (LTO); and
- Lithium-iron phosphate (LFP).

A recent report by California's Air Resources Board (CARB 2016) on battery costs for heavy-duty electric vehicles confirms that NMC is the lithium ion battery chemistry used by New Flyer, as well as by Proterra, one of their electric bus competitors, on some of their models too. A cost for NMC battery packs is also indicated, in the range of $750 to $850 per kWh (USD). Based on the common battery pack size of 200 kWh, as employed in the Altoona-tested New Flyer XE40 bus, and using standard currency conversion, this translates to a total cost range of $188,000 to $212,000 per bus for batteries. An earlier report (CARB 2015) confirms the specific energy of NMC battery packs is typically around 0.15 kWh per kg, such that the overall weight of the same common battery pack is about 1,330 kg.

In terms of composition, the four main minerals contained in NMC batteries include: lithium, cobalt, nickel and manganese. Despite "lithium" as a common connection, there is surprisingly little present within any type of lithium ion battery, with this also primarily involved in the electrolyte. A recent minerals industry story suggests in practical terms lithium content is approximately 0.16 kg per kWh (Industrial Minerals 2016). As such, for a 200 kWh NMC battery pack this translates to approximately 32 kg of lithium, or just over roughly 2% of overall pack mass, a relatively small amount.

In terms of the other three minerals, cobalt is the most critical, being used primarily at the cathode. As noted by Petersen (2016), typical active cathode mass is around 2.4 kg per kWh, and for NMC chemistry, cobalt represents approximately 15% of the cathode, or about 0.36 kg per kWh. As further noted by Patel (2015), the NMC cathode today involves typically equal parts by weight of nickel, manganese and cobalt. As such, for a 200 kWh pack, the content of each of these minerals translates to approximately 72 kg, with the three cathode materials combined representing about 16% of overall pack mass.

Nitta et al. (2015), as part of an overview analysis of materials involved in lithium-ion batteries, includes rough five-year price ranges for individual minerals. Their brief analysis shows lithium and cobalt are the most expensive of the four in recent times, with cobalt also being the most variable. Of the four, manganese is the most generally abundant, with a significantly lower price, i.e., order of magnitude lower, such that it is unlikely to be a concern. Nickel also has been relatively stable, again less of a concern. Further consideration is thus given primarily to cobalt and secondarily to lithium.

Historical average annual prices for cobalt have been consistently tracked by the US Geological Service (Shedd 2013), with these, as converted, presented in **Figure 5** for the years 1970 through 2010. Cobalt, as illustrated, has been a relatively volatile mineral commodity over time. Added to **Figure 5** are approximated values for more recent years, with these based on available data from a commodity trading website (Trading Economics 2017), and covering 2012 to present. From 2012 through 2016 cobalt had been relatively stable, but most recently, over a one-year period, from 2016 to 2017, cobalt escalated rapidly, by roughly 142%. It is important to note that compared to historical pricing, the current peak is actually not the highest spike price experienced over the past 50 years. This rapid increase, however, illustrates the vulnerability of batteries to mineral price escalations.

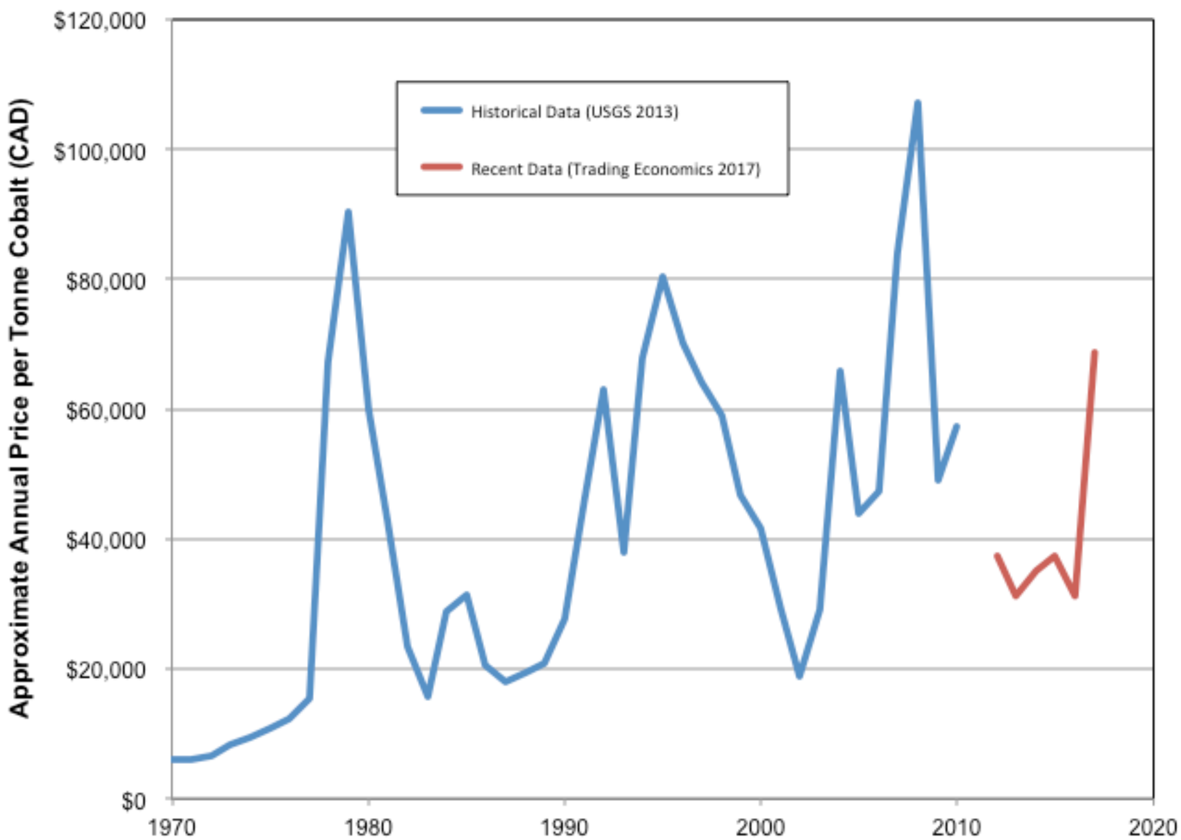

Figure 5. Historical average price of cobalt per tonne, translated (data from Shedd 2013 and Trading Economics 2017)

For this analysis, impacts are assessed based on the assumption of a continued annual price escalation of approximately 150% for cobalt, roughly matching the trend over the past year. Assuming a current price corresponding to approximately $75,000 per tonne as the base for calculation, the estimated additional cost associated with cobalt for a further 150% price increase and use of 72 kg per pack, translates to approximately $8,100.

The increasing trend in lithium prices over the recent past has been relatively predictable (Trading Economics 2017), representing approximately a 25% increase annually. Assuming a current price corresponding to approximately $25,000 per tonne as a base for calculation, the estimated cost associated with lithium for a further 25% price increase and use of 32 kg per pack, translates to approximately $200, much less than that for cobalt.

Together, continued price escalations of cobalt and lithium represent a potential increased cost of approximately $8,300 per bus, or roughly a 4% increase for the NMC battery pack, based on the current cost range. These impacts directly affect the purchase cost of the bus, and as such represent directly present values. Further, unlike other component issues being considered, the annual travel of the bus has no effect in this case.

Cobalt is clearly the dominant concern (West 2016). An important consideration in this regard is further technology improvements that could reduce dependency and price impacts. Already, it is well known that the NCA battery chemistry, which for example is already used in certain Tesla products, requires roughly 40% less cobalt, and that LFP and LMO chemistries require negligible cobalt. Further improvements in performance and new, less costly battery technologies under development (Nitta et al. 2015) could also significantly reduce impacts and dependency on cobalt.

Concerns with cobalt also provide a relevant focus for CSR. Part of the problem with cobalt is that a dramatically high proportion of the mineral resides in the Democratic Republic of the Congo. Indeed, the most significant spikes in past cobalt prices have been primarily due to instability and problems associated with this country or its predecessors (Shedd 2013). At the same time, concerns with exploitative practices have been raised regarding cobalt, as for example highlighted in a number of expose articles in the Washington Post (Frankel 2016) and Financial Times (Pilling 2017), as well as a recent Amnesty Interactional (2016) report related to cobalt.

Potential costs associated with CSR externalities from cobalt have not been evaluated in this report, given they are very difficult to precisely determine and to differentiate from other effects. At the same time, this issue is crucial to identify. Into the future, battery manufacturers will need to pay specific attention to working conditions and codes of practice for sub-suppliers providing minerals from developing countries, in particular locations like the Democratic Republic of the Congo.

8. Weight-Induced Infrastructure Damage

A subtle issue involved with the transition to electric buses, is the potential for increased damage to roadway infrastructure caused by their higher weight compared to conventional diesel buses. This certainly involves a negative implication for electric buses, but needs to be considered in context. Compared to other factors investigated, the monetized values, as calculated, are notable but are not overwhelming. Further, buses contribute only a small proportion of roadway damage overall in any event, a situation that would not change appreciably with any significant transition to electric buses. Lastly, within Winnipeg, transit buses, whether diesel or electric powered, have historically been permitted to exceed vehicle weight limits, as normally imposed under By-Law 1573/77 (George 2015). From a legal perspective, this makes bus weight a moot point. Incremental damage costs are still determined in order to obtain a full understanding of all implications.

Determining damage to roadways can be difficult in the absence of active traffic and infrastructure monitoring data. An important approach for such analysis has been to use the measure of Equivalent Single Axle Load (ESAL), developed by the American Association of State Highway Officials (AASHO), to approximate contributed damage. ESAL is a form of load equivalency factor approximated by considering the estimated load on each vehicle axle in pounds (lb), dividing by a standard reference value for each axle of 18,000 lb, which is equivalent to approximately 80 kN, and then taking the fourth power of the resulting ratio value. Some further adjustment can be made to account for dual set of wheels where present on a particular axle. For a vehicle being considered, the ESAL values for all axles are finally summed to obtain an overall result. Similarly, roadways can be assigned aggregate ESAL ratings based on the nature of construction, i.e., thickness of roadway layer.

Recently, the American Public Transit Association looked at this approach as applied to transit systems, using technical expertise based in Winnipeg (MORR/CUTR 2014). Fortuitously, similar use of ESAL to evaluate transit-based roadway damage was recently examined using Winnipeg as a case study (George 2015), specifically considering Winnipeg Transit's #162 Route, an express route travelling from downtown to the University of Manitoba, in part along the Southwest Rapid-Transit corridor. The analysis and data from this latter work are used as the basis for evaluation here.

An analysis of vehicle-km (VKM) travel within the City of Winnipeg is available for the three main types of vehicles involved, based on the following sources: light duty vehicles (Transport Canada 2016); freight vehicles (Transport Canada 2016); and transit buses (City of Winnipeg ND). Together with ESAL approximations for each type, the estimated overall damage contributions by type of vehicle are estimated as follows, based on total ESAL-VKM values:

- Approximately 95.88% of overall damage is caused by freight vehicles;
- Approximately 3.85% of overall damage is caused by transit buses; and
- Approximately 0.27% of overall damage is caused by light-duty passenger vehicles.

A model bus route is considered in the analysis, based on a one-way route length of 26.7 km. In terms of ESAL rating, the bus route is assumed to consist of a mix involving 96% "highway" and 4% "arterial road." The highway component, with thickness of 250+ mm, has a rough ESAL rating of 15,000,000, while the arterial road component, with 175-225 mm thickness, has a rough ESAL rating of 4,000,000. The resulting approximate overall ESAL rating is thus 10,940,000 (Canadian Strategic Highway Research Program 2001, Iowa Statewide Urban Design and Specifications 2013, ENG-TECH Consulting 2016). Further, in term of estimating affected roadway area, the highway component involves 4 lanes, while the arterial road component involves 2 lanes, each 3.7 m wide. Assuming a 20-year roadway repair cycle for the model route, approximately 5% of the surface area undergoes rehabilitation or reconstruction each year, representing for the model route a total affected area of 18,900 m2.

In terms of associated repair costs, data from the City of Winnipeg (2017) are used. Combined with the further assumption of a split of 80% rehabilitation versus 20% reconstruction, roadway repair costs translate to a total of approximately $326.45 per m2. This is attributable to damage caused by all modes of transportation. For the model bus route, this translates to annual costs of about $6,170,000. The proportion of damage attributable specifically to transit buses (i.e., 3.85%) translates to $12.55 per m2 or about $237,000 per year.

Finally, ESAL values indicating comparative roadway damage for the electric versus diesel buses are based on public data, described earlier, for the almost identical New Flyer XE40 and XD40 buses respectively (Pennsylvania State University 2015 and 2012). A series of standard weight measures are provided for the two types of buses, with in general terms, the electric bus version being somewhat heavier, i.e., in the range of 2,000 lb to 5,500 lb heavier depending on the specific measure considered. It is important to note that, although both types of buses have front axle loads that are easily less than front axle restriction values, both the electric and diesel buses can readily exceed rear axle and total weight restriction values as outlined under Winnipeg's By-Law 1573/77.

Annual travel per bus is an important variable as discussed earlier. In this case, variations in annual travel are translated to a higher or lower number of circuits of the model route per day. The last important factor in evaluating bus weight is passenger loading. As described earlier, using average loading data from APTA, it is assumed that each bus has an average of 10 passengers at all times. This represents a load factor of 13.2% based on total capacity for the electric bus (seated and standees). Although not used in calculations, given the higher number of seats on the electric bus version, the capacity factor based only on seated load is approximately 26%. A standard average passenger weight of 150 lb is also assumed.

Based on this consistent passenger loading, the total weight and ESAL values for the two types of buses are estimated as follows:

- Electric bus weighing 34,200 lb, with ESAL value of 1.73; and
- Diesel bus weighing 27,300 lb, with ESAL value of 0.94.

From these values, combined with annual travel, the incremental overall increase in ESAL-based damage contributions by buses for the model route are calculated as follows along with annual cost, based on the three travel cases:

- **Lower annual travel**: 0.0095% increase in damage by buses to model route, or $600 per year
- **Average annual travel**: 0.0135% increase in damage by buses to model route, or $856 per year

- **Higher annual travel**: 0.0190% increase in damage by buses to model route, or $1,200 per year

Lastly these annual costs are translated to present values as follows:

- **Lower annual travel**: $600 per year × 9.20 PVIFA = $5,520 present value
- **Average annual travel**: $856 per year × 9.20 PVIFA = $7,880 present value
- **Higher annual travel**: $1,200 per year × 9.20 PVIFA = $11,040 present value

These values are notable, but are not overwhelming compared to other component issues.

9. Photochemical Smog Precursor Emissions

The reduction of photochemical smog precursor emissions, specifically involving nitrogen oxides (NOx) and volatile organic compounds (VOC), provides a positive contribution for the transition from diesel buses to electric transit buses. The monetized values involved are material but not high. Importantly, results show that costs for NOx emissions are by far the most significant of those associated with four interrelated air pollutants: NOx and VOC, described in this section; particulate matter, described later (see page 25); and air toxics, also described later (see page 29). This matches the results of recent reporting by Health Canada (2016) showing NOx levels within Manitoba, particularly the vicinity of Winnipeg, to be relatively high on a national basis.

Photochemical smog has become a common health and environmental scourge for urban metropolitan areas. It involves a complex of reactions occurring in the atmospheric space above cities and industrial areas. It is responsible for a variety of identified problems, including human health and environmental degradation. The main precursor emission constituents, also termed “primary pollutants,” involve two main groups:

- Nitrogen oxides (NOx), primarily including nitrogen oxide (NO) and nitrogen dioxide (NO2). A third constituent oxide, nitrous oxide (N2O) is an identified GHG, but generally is not involved to any great extent with photochemical smog. Nitrogen oxides are damaging in themselves, but also involved in the atmospheric formation of additional problematic constituents. They can also, as described in a later section (see page 34), separately contribute to acid precipitation.
- Volatile organic compounds (VOC), often considered in terms of non-methane volatile organic compounds (NMVOC). These are primarily involved in the atmospheric formation of additional problematic constituents. Some can also exert potentially carcinogenic effects, with those implications discussed separately in the section on air toxics (see page 29).

Both these groups are identified as “criteria air contaminants” (EPA 2015). From data presented in Cai et al. (2013), the emission factors for releases of these constituents from modern diesel bus vehicles are approximately as follows:

- NOx emissions of 0.756 g per km. Based on current understanding, most NOx immediately after exhaust still consists primarily of NO, although NOx in modern diesel engines can have a relatively significant proportion of NO2. Also, over time, NO is converted to NO2 (refer to During et al. 2011).
- VOC emissions of 0.054 g per km.

Based on these values, the net higher emissions of NOx and VOC associated with diesel buses compared to electric on an annual basis are summarized as follows:

- **Lower annual travel**: 26.5 kg NOx and 1.9 kg VOC per year net for diesel bus
- **Average annual travel**: 37.8 kg NOx and 2.7 kg VOC per year net for diesel bus
- **Higher annual travel**: 53.0 kg NOx and 3.8 kg VOC per year net for diesel bus

There are two main problematic constituents that are formed in the atmosphere above urban areas as part of photochemical smog, which are termed "secondary pollutants." These are:

- Low-level ozone (O3). Ozone in the high atmosphere has been long identified as essential to protecting all life forms on Earth from ultraviolet radiation, however, when ozone occurs at low levels it becomes a particularly problematic and damaging oxidant, which is directly linked to a variety of health and environmental concerns.
- Peroxyacetyl nitrate (PAN) and related constituents. PAN is a known irritant and is also known in particular to be highly toxic to plant life (Kruus and Valeriote 1984).

The atmospheric reactions involved with photochemical smog are highly complex, as noted, but their basics have been understood for some time (Wark and Warner 1976). Part of the complexity is given that the formation of low-level ozone can, depending on circumstances, be controlled either by NOx or by VOC as the limiting factor. As noted by NAS (1991), if the atmospheric ratio of VOC:NOx is higher than a critical level, i.e., in the range of 6:1 to 9:1, the NOx becomes the limiting factor and controls ozone formation. Below that approximate ratio, NOx is effectively saturated, with the formation of ozone being controlled by VOC.

The ratio of VOC:NOx, based on the emission factors cited above, and also reflecting conditions immediately at exhaust, is approximately 0.07:1 (note mass basis), meaning that NOx is saturated and VOC controls ozone formation. Checking relevant air quality data that has been published for Winnipeg during the early 2000s confirms the same situation, with the VOC:NOx ratio being very low (Krawchuk and Snitowski 2008). As such, for the conditions associated with a diesel bus in Winnipeg it is reasonable to assume that VOC is indeed the controlling factor for ozone formation. Based on this, it is assumed for simplicity that ozone impacts, discussed later, can be allocated solely on the basis of VOC emissions.

In terms of damage costs, Health Canada (2016) recently conducted a health risk assessment of diesel exhaust, and estimated a variety of health related costs. They used 2015 as the based years for evaluation, and showed that, for Canada as a whole, premature mortalities alone from on-road diesel exhaust result in a cost of roughly $2.3 billion, specifically noting a per-mortality cost of \$6.5 million. In their report, they identified costs and adverse health-outcomes to be caused by three pollutants: particulate matter (discussed later, see 25), nitrogen dioxide (NO2) and low-level ozone (O3). The latter two are most relevant in terms of photochemical smog related pollutants. Further, given that transit buses are operated on-road and largely in an urban setting, costs associated with plant damage, in particular to agriculture (lost production yields for crops) and visibility in non-urban areas (haze), are assumed to be negligible.

Analysis of premature mortalities specifically for Manitoba in 2015 by Health Canada (2016) shows approximately 14 premature deaths associated with NO2 for on- and off-road combined. Costs associated with premature mortalities related to NO2 are assumed to be appropriately apportioned based on NOx emissions. Total NOx emissions for Manitoba in that year are identified in the same report as 14,149,000 kg, including both on- and off-road. (Note although premature mortalities are provided for on-road only within Manitoba, data on overall NOx emissions for Manitoba are only provided for combined on- and off-road). Based on the reductions per year for each electric bus, as noted above, annual savings per electric bus translate as follows:

- **Lower annual travel**: 26.5 kg NOx ÷ 14,149,000 kg NOx × 14 premature deaths × \$6,500,000 = \$170.44 per year
- **Average annual travel**: 37.8 kg NOx ÷ 14,149,000 kg NOx × 14 premature deaths × \$6,500,000 = \$243.11 per year
- **Higher annual travel**: 53.0 kg NOx ÷ 14,149,000 kg NOx × 14 premature deaths × \$6,500,000 = \$340.88 per year

Analysis of premature mortalities specifically for Manitoba in 2015 by Health Canada (2016) also shows approximately 4 premature deaths associated with ozone for on- and off-road combined. Costs associated with premature mortalities related to ozone are assumed to be appropriately apportioned based on VOC emissions. Total VOC emissions for Manitoba in that year are identified in the same report as 1,139,000 kg, including both on- and off-road. (Note although premature mortalities are provided for on-road only within Manitoba, data on overall VOC emissions for Manitoba are only provided for combined on- and off-road). Based on the reductions per year for each electric bus, as noted above, annual savings per electric bus translate as follows:

- **Lower annual travel**: 1.9 kg VOC ÷ 1,139,000 kg VOC × 4 premature deaths × \$6,500,000 = \$43.37 per year
- **Average annual travel**: 2.7 kg VOC ÷ 1,139,000 kg VOC × 4 premature deaths × \$6,500,000 = \$61.63 per year
- **Higher annual travel**: 3.8 kg VOC ÷ 1,139,000 kg VOC × 4 premature deaths × \$6,500,000 = \$86.74 per year

The same report identifies additional (non-death related) adverse health outcome occurrences resulting from ozone exposure associated with on-road diesel exhaust, but without assigning specific costs. These adverse impacts include increased numbers of acute respiratory symptom days, restricted activity days, asthma symptom days, hospital admissions, emergency room visits, child acute bronchitis episodes and adult chronic bronchitis cases across Canada. At the same time, relevant cost data for such adverse health outcomes are presented in Weisbrod et al. (2008). The unit cost data outlined by them are originally from Canadian sources but are dated (i.e., from 1999). The data set included an average premature mortality cost of $2.48 million at that time, which when compared to the 2016 value of $6.5 million, suggests a 160% increase over 17 years. From a financial analysis perspective, on a year-by-year basis, this translates to an approximate 5% increase per year, which is reasonable. Using the identified occurrences and unit costs for adverse health outcomes, total costs for ozone translate to approximately $15.5 million for Canada as a whole during 2015. As noted earlier, costs associated with these other adverse health incomes are assumed to be appropriately apportioned based on VOC emissions. Total on-road VOC emissions for Canada in that year are identified in the same report as 37,992,000 kg, Based on the reductions per year for each electric bus, as noted above, annual savings per electric bus translate as follows:

- **Lower annual travel**: 1.9 kg VOC ÷ 37,992,000 kg VOC × $15,500,000 = $0.78 per year
- **Average annual travel**: 2.7 kg VOC ÷ 37,992,000 kg VOC × $15,500,000 = $1.10 per year
- **Higher annual travel**: 3.8 kg VOC ÷ 37,992,000 kg VOC × $15,500,000 = $1.56 per year

Combining the annual values for both nitrogen dioxide and ozone related costs, and multiplying by PVIFA value, the present values are as follows:

- **Lower annual travel**: ($170.44 + $43.37 + $0.78) per year × 9.20 PVIFA = $1,970 present value
- **Average annual travel**: ($243.11 + $61.63 + $1.10) per year × 9.20 PVIFA = $2,810 present value
- **Higher annual travel**: ($340.88 + $86.74 + $1.56) per year × 9.20 PVIFA = $3,940 present value

These values are material, but are not overly high. The results also confirm that savings are predominantly associated with NOx reductions rather than VOC reductions.

10. Particulate Matter Emissions

The reduction of particulate matter (PM) emissions provides a positive contribution for the transition from diesel to electric transit buses. The monetized values involved, however, are relatively small. At the same time, PM is a subtly impactful air pollutant that has begun to be viewed as increasingly important, especially in terms of health effects.

Particulate matter, as suggested by its name, involves relatively fine particles of solids or liquid droplets that are sufficiently small that they tend to stay airborne for relatively long periods of time (Wark and Warner 1976). Coarser materials, greater than about 500 μm (micron) or 0.5 mm effective diameter, tend to drop out of the air relatively quickly. More conventionally, the typically measured-parameter has been PM10, or particles less than 10 µm (micron) effective diameter. More recently, the shift of concern has been toward fine particulate matter, considered as PM2.5, i.e., particles less than 2.5 µm (micron) effective diameter, and ultra-fine-particulate (UFP) matter. The small sizes involved with PM2.5 and UFP makes human exposure to them of particular concern, given they are highly respirable and thus able to reach deep into the lungs (EPA 2002).

From data presented in Cai et al. (2013), the emission factors for releases from modern diesel bus vehicles are approximately 0.0128 g per km for PM10 and 0.0124 g per km for PM2.5. These figures illustrate a known concern with diesel combustion, with virtually all the particulate emissions being fine in nature, i.e., almost 97%. Cai et al. (2013) also include emission factors for PM10 and PM2.5 resulting from tire and brake wear on buses. However, these are virtually the same for both types of buses, and are not considered further.

Diesel fuel used in diesel cycle engines has been commonly known to produce smoke and soot under certain conditions (Lilly 1984). As such, diesel vehicles have been considered as much more of a concern regarding PM emissions. In other data presented by Cai et al. (2013), the emission factors of PM10 and PM2.5 for a typical modern passenger vehicle using gasoline as fuel are both respectively about one-third per km that for an individual diesel bus. Although certainly higher for an individual diesel bus, the extent of PM emissions is not as relatively high as found for GHG emissions.

The most important emission measure is PM2.5. Based on emission factor values as presented, along with the further assumption that diesel fuel used for auxiliary heating is catalytically combusted with negligible PM, net higher emissions of particulates associated with diesel buses compared to electric are summarized as follows:

- **Lower annual travel**: 0.434 kg PM2.5 per year net for diesel bus
- **Average annual travel**: 0.620 kg PM2.5 per year net for diesel bus
- **Higher annual travel**: 0.868 kg PM2.5 per year net for diesel bus

The very large surface area of fine particulate, i.e., PM2.5, also means that these particles present a very high adsorption potential for chemical species that may be formed as by-products of diesel combustion and also may be carcinogenic. As such, PM2.5 has been associated with both adverse respiratory and carcinogenic effects, although for the latter PM acts largely as a vector. For the purpose of analysis, the carcinogenic component of effects is included separately along with air toxics (see page 29). Further, given that transit buses are operated on-road and largely in urban settings, costs associated with emission damage to plants, in particular to agriculture (e.g., lost production yields for crops), and visibility in non-urban areas (e.g., haze), are not included.

Similar to nitrogen oxides, as described in the last section, particulate matter is associated with premature mortalities. For Manitoba, Health Canada (2016) estimated these to represent a total of 15 cases in 2015, including both on- and off-road. Of this number, roughly 20% are associated with cancer, and included later under air toxics (see page 29), with the remaining deaths caused by three major adverse conditions in order: chronic exposure ischemic heart disease (IHD), chronic exposure chronic obstructive pulmonary disease (COPD), and chronic exposure cerebrovascular mortality. Damage associated with these premature mortalities is assumed to be apportioned based on PM2.5 emissions. For 2015, PM2.5 emissions in Manitoba totaled 697,000 kg as noted in the same report, again including both on- and off-road. (Note although premature mortalities are provided for on-road only within Manitoba, data on overall PM2.5 emissions for Manitoba are only provided for combined on- and off-road). Based on the reductions per year per bus, annual savings per electric bus translate as follows:

- **Lower annual travel**: 0.434 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.80 non-cancer × $6,500,000 = $48.57 per year
- **Average annual travel**: 0.620 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.80 non-cancer × $6,500,000 = $69.38 per year
- **Higher annual travel**: 0.868 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.80 non-cancer × $6,500,000 = $97.14 per year

The same report identified additional, non-death, adverse health outcomes resulting from PM2.5 exposure associated with on-road diesel exhaust, but without assigning specific costs. (Note that total on-road PM2.5 emissions are provided for Canada). These adverse impacts include increased numbers of acute respiratory symptom days, restricted activity days, asthma symptom days, hospital admissions, emergency room visits, child acute bronchitis episodes and adult chronic bronchitis cases across Canada. At the same time, relevant cost data for such adverse health outcomes are presented in Weisbrod et al. (2008) and adjusted as described in the last section.

Using the identified occurrences and updated costs for adverse health outcomes, these total costs translate to approximately $237 million for Canada as a whole in 2015. Costs associated with these other adverse health incomes are assumed to be appropriately apportioned based on PM2.5 emissions on a national basis. Total on-road PM2.5 emissions for Canada in that year are identified in the same report as 24,604,000 kg, Based on the reductions per year per bus, as noted above, annual savings per electric bus translate as follows:

- **Lower annual travel**: 0.434 kg PM2.5 ÷ 24,604,000 kg PM2.5 × $237,000,000 = $4.18 per year
- **Average annual travel**: 0.620 kg PM2.5 ÷ 24,604,000 kg PM2.5 × $237,000,000 = $5.97 per year
- **Higher annual travel**: 0.868 kg PM2.5 ÷ 24,604,000 kg PM2.5 × $237,000,000 = $8.36 per year

Combining the annual values for both premature mortalities and other adverse health outcome-related costs, and multiplying by PVIFA value, the present values are as follows:

- **Lower annual travel**: ($48.57 + $4.18) per year × 9.20 PVIFA = $480 present value

- **Average annual travel**: (\$69.38 + \$5.97) per year × 9.20 PVIFA = \$690 present value
- **Higher annual travel**: (\$97.14 + \$8.36) per year × 9.20 PVIFA = \$960 present value

These values are notable but are not high. The results also clearly show that in terms of relevant air pollutants, particulate matter reductions are relatively less important in Winnipeg that NOx emission reductions.

11. Carbon Monoxide Emissions

The reduction of carbon monoxide emissions provides a positive contribution for the transition from diesel to electric transit buses, however the monetized values involved are relatively small. At the same time, an interesting future opportunity is also identified for transit bus garages once a complete transition to electric is achieved. Ventilation requirements, which can be costly depending on climate, are primarily dictated today by the potential presence of carbon monoxide, and could be significantly reduced if diesel buses are totally converted to electric.

Carbon monoxide is classified as a “criteria air contaminant” (EPA 2015) and is thus important to consider. However, it exhibits characteristics that are somewhat unexpected for such a pollutant (Bailey et al. 1978). Carbon monoxide is normally present in the atmosphere in low concentration, around 10×10^{-6} % by volume, but with these normal levels being generally higher than other identified criteria air pollutants. The vast majority of worldwide CO emissions are known to be from natural sources, yet even given the magnitude of releases, the overall CO level remains relatively low and stable, indicating that CO clears quickly, albeit in ways that are not entirely understood. Despite its close structural similarity to carbon dioxide, CO does not act as a GHG. Further, oxidation of CO within the atmosphere to CO2 is known to be very slow, such that CO does not contribute to direct GHG formation, although its presence in excess is known to interfere with the breakdown and clearing of methane from the atmosphere (Jardine et al. 2004). The latter is a potent GHG, such that this effect can indirectly contribute.

Levels of CO in urban areas are generally much higher than background. Importantly, the largest human-caused source of CO emissions is from motor vehicles, it being formed as a result of incomplete combustion. Although a variety of subtle adverse effects may be associated with CO, the primary concern is its acute toxicity (WHO 1999), particularly in confined spaces. Carbon monoxide binds selectively and strongly with haemoglobin in the blood, blocking oxygen. Discomfort and neurological impairment can occur at concentrations as low as 100 ppm, while exposure to concentrations of roughly 1,000 ppm, depending on exposure time, can lead to death. As the concentration increases, the risk of death also increases. Yet despite its acute toxicity, the effects of CO can be to a large extent rapidly reversed if a person is removed quickly from the exposure threat.

From data presented in Cai et al. (2013), the emission factor for carbon monoxide release from modern diesel bus vehicles is approximately 0.68 g per km. Based on this and the further assumption that diesel fuel used for auxiliary heating is catalytically combusted with negligible

CO, net higher emissions of CO associated with diesel buses compared to electric are summarized as follows:

- **Lower annual travel**: 23.8 kg CO per year net for diesel bus
- **Average annual travel**: 34.0 kg CO per year net for diesel bus
- **Higher annual travel**: 47.6 kg CO per year net for diesel bus

The damage costs associated with carbon monoxide emissions have been estimated in a variety of past studies, but do not appear to be addressed in more recent literature. In all cases, associated costs are not overly high, in the rough range of $200 to $1,000 per tonne. The direct accessibility of original older work is a problem, with values from many older studies being only indirectly available. Work by Matthews et al. (2001) provides a reasonable cost for CO of approximately $520 per ton (USD). Converting units and currency, this translates to a reasonable value of about $710 per tonne here.

This combined with a PVIFA value of 9.20, provides the following total monetized present value impacts:

- **Lower annual travel**: 23.8 kg CO × $710 per tonne × 9.20 PVIFA = $160 present value
- **Average annual travel**: 34.0 kg CO × $710 per tonne × 9.20 PVIFA = $220 present value
- **Higher annual travel**: 47.6 kg CO × $710 per tonne × 9.20 PVIFA = $320 present value

As seen from results, the reduction of CO in relative terms by an electric bus is significant, i.e., effective elimination. At the same time, the monetized savings in absolute terms are not large. The elimination of CO emissions does present into the future a prospective opportunity. Garages for diesel buses require significant and adequate ventilation, primarily dictated by the potential presence of carbon monoxide. This includes significant costs for ventilation air heating during the winter. If carbon monoxide is eliminated, the need for ventilation is drastically reduced, with associated savings. However, achieving such a result requires total conversion of a transit bus fleet, with results being negligible if only an initial small number of buses are electric. Further analysis would be required to estimate the value of such savings, and they nevertheless remain well into the future.

12. Air Toxics Emission

The reduction of air toxics emissions provides a positive contribution for the transition from diesel to electric transit buses. The monetized values involved, however, are quite small. At the same time, the results also appear to confirm that cancer risks associated with diesel exhaust remain relatively small.

The term “air toxics” was defined in the U.S. to include any of 187 specific chemical compounds and constituents identified as hazardous air pollutants under the U.S. Clean Air Act. This term is not commonly used within Canada for official purposes, and, as such, health risk assessments of diesel exhaust here, such as Health Canada (2016), do not explicitly include this category of impacts. Nevertheless, the same constituents are also identified under the Canadian Environmental Protection Act within the category of toxic substances, and are still a relevant concern. For this analysis, air toxics are considered primarily in terms of cancer risk impacts and costs.

The divergent situation between Canada and the U.S. appears to reflect a difference in national attitude toward cancer and cancer risks, with these much more strongly emphasized in the U.S. Hazardous air pollutants are frequently associated with cancer, with many constituents specifically identified as carcinogenic. Diesel exhaust contains many potential carcinogens, and, indeed, in 2012 the International Agency for Research on Cancer declared that diesel exhaust itself is considered a carcinogen (IARC 2012 with detailed follow-up in IARC 2013). As noted in an earlier section, Health Canada (2016) identified cancer-related impacts associated with PM2.5, although with the particulates really acting more as a vector, i.e., adsorbing other carcinogenic chemicals and carrying them deep into the lungs. These impacts of PM2.5 are included here.

In the report by Health Canada (2016) 20% of the 15 premature mortalities associated with PM2.5 identified for Manitoba are related to cancer, specifically due to chronic-exposure lung cancer. As noted in an earlier section, these are apportioned based on PM2.5 reductions per bus compared to PM2.5 emissions for the province as a whole. Costs for this translate as follows:

- **Lower annual travel**: 0.434 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.20 cancerous × $6,500,000 = $12.14 per year
- **Average annual travel**: 0.620 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.20 cancerous × $6,500,000 = $17.35 per year
- **Higher annual travel**: 0.868 kg PM2.5 ÷ 697,000 kg PM2.5 × 15 premature deaths × 0.20 cancerous × $6,500,000 = $24.28 per year

Individual chemical constituents that are hazardous air pollutants have been recognized for some time to be present in the exhaust from diesel engines (Lilly 1984). Beyond PM2.5, however, standardized emissions factors for hazardous air pollutants have not been developed. Nevertheless, It is still possible to approximate releases and costs for the two most important such constituents, which are formaldehyde and benzene.

As outlined by McClellan et al. (2012), three aggregate categories of chemical constituents are prominent within diesel exhaust, in terms of hazardous air pollutants. These categories are as follows, in declining order of output from a standardized diesel test engine setup:

- Carbonyl compounds, including formaldehyde, acetaldehyde, acetone, acrolein and others, together representing approximately 255 mg per hour, which based on other data presented by them involves about 75% as formaldehyde itself;

- Single-ring aromatics, including benzene, toluene, ethyl-benzene, xylene, and other related compounds, together representing approximately 72 mg per hour, which based on other data presented by them involves only about 1.5% as benzene itself; and
- Polycyclic aromatic hydrocarbons (PAH), involving various constituents, together representing approximately 70 mg per hour.

Zhou et al. (2015) undertook an important review of cancer-related hazardous air pollutants for the U.S. that very quickly simplifies analysis. They found, based on 2005 emissions data, that the cumulative annual cancer-related health impacts of inhaling the top 10 carcinogenic hazardous air pollutants represent about 1,600 disability adjusted life-years (DALY) for the U.S. as a whole or about 0.60 DALYs per 100,000 people. Further, their results show the top two alone, formaldehyde and benzene, represent close to 60% of this, roughly 900 DALYs for the U.S. as a whole, or 0.33 DALYs per 100,000 people. As discussed, both formaldehyde and benzene are present in diesel exhaust.

Formaldehyde emissions from a diesel engine, based on McClellan et al. (2012), represent approximately 191 mg per hour (i.e., 255 mg per hour × 0.75). Formaldehyde emissions in the U.S. as a whole for 2005 are approximately 10,000,000 kg (EPA 2007), and, based on Zhou et al. (2015), represent 480 DALYs. Given emissions on an hourly rather than per km basis, an average speed of 20 km per hour is assumed, as discussed earlier. The further assumption is made that there are effectively no formaldehyde emissions from an electric bus, such that the net annual reductions compared to a diesel bus are as follows, expressed as DALY:

- **Lower annual travel**: 35,000 km ÷ 20 km per hour × 191 mg per hour ÷ 10,000,000 kg × 480 DALY = 16.1 × 10^{-6} DALY on an annual basis
- **Average annual travel**: 50,000 km ÷ 20 km per hour × 191 mg per hour ÷ 10,000,000 kg × 480 DALY = 22.9 × 10^{-6} DALY on an annual basis
- **Higher annual travel**: 50,000 km ÷ 20 km per hour × 191 mg per hour ÷ 10,000,000 kg × 480 DALY = 32.2 × 10^{-6} DALY on an annual basis

Benzene emissions from a diesel engine, based on McClellan et al. (2012), represent approximately 1.1 mg per hour (i.e., 72 mg per hour × 0.015). Benzene emissions in the U.S. as a whole for 2005 are approximately 2,730,000 kg annually (EPA 2009), and, based on Zhou et al. (2015), represent 400 DALYs. Given emissions on an hourly rather than per km basis, an average speed of 20 km per hour is assumed, as also used earlier in the section on noise (see page 13). Further assuming effectively no benzene emissions from an electric bus, the net annual reductions compared to a diesel bus are as follows, expressed as DALY:

- **Lower annual travel**: 35,000 km ÷ 20 km per hour × 1.1 mg per hour ÷ 2,730,000 kg × 400 DALY = 0.29 × 10^{-6} DALY on an annual basis
- **Average annual travel**: 50,000 km ÷ 20 km per hour × 1.1 mg per hour ÷ 2,730,000 kg × 400 DALY = 0.44 × 10^{-6} DALY on an annual basis
- **Higher annual travel**: 70,000 km ÷ 20 km per hour × 1.1 mg per hour ÷ 2,730,000 kg × 400 DALY = 0.58 × 10^{-6} DALY on an annual basis

An economic value of approximately $110,000 per DALY was selected for analysis purposes. This value was noted recently for a Canada application, in a study by Orenstein et al. (2010) for Alberta Health Services. Their work looked specifically at economic costs associated with occupational-related cancers. Adding these major cancer impacts together, translates to the following annual costs:

- **Lower annual travel**: (16.1 + 0.29) × 10-6 × $110,000 = $1.80 per year
- **Average annual travel**: (22.9 + 0.44) × 10-6 × $110,000 = $2.57 per year
- **Higher annual travel**: (32.2 + 0.58) × 10-6 × $110,000 = $3.60 per year

Combining the annual values for PM2.5, formaldehyde and benzene related cancer costs, and multiplying by PVIFA value, the present values are as follows:

- **Lower annual travel**: ($12.14 + $1.80) per year × 9.20 PVIFA = $130 present value
- **Average annual travel**: ($17.35 + $2.57) per year × 9.20 PVIFA = $180 present value
- **Higher annual travel**: ($24.28 + $3.60) per year × 9.20 PVIFA = $260 present value

The calculated values are all quite small. Importantly, these results also confirm that cancer risks associated with diesel exhaust appear to be very small.

13. Used Diesel Engine Lubrication Oil Disposal

The reduction of used lubrication oil, which needs disposal, provides a positive contribution for the transition from diesel buses to electric transit buses. The monetized values involved are overall relatively small. Waste oil could represent a significant environmental problem, however, this issue appears to have been largely addressed already through the implementation of appropriate stewardship programs across the country.

All internal combustion engines, including diesel engines, require the use of lubricating oils for sustained proper operation (Lilly 1984). This also means that all diesel engines periodically generate quantities of used (or waste) lubricating oil. These must be disposed, and, as such, impose a cost. The vast majority of lubrication oils are specialized hydrocarbon oil products made from refined petroleum sources.

For analysis purposes, the advanced Cummins ISL diesel engine is assumed, which has been employed for New Flyer's Xcelsior Model XD-40 buses. As outlined by Cummins Inc. (2008), this engine involves a recommended oil change interval every approximately 10,000 km, with a total oil volume of 27.6 Litres. Based on this, annual used-oil generation is as follows for the assumed annual travel cases:

- **Lower annual travel**: 35,000 km ÷ 10,000 km per change × 27.6 Litres = 96.6 Litres per year

- **Average annual travel**: 50,000 km ÷ 10,000 km per change × 27.6 Litres = 138.0 Litres per year
- **Higher annual travel**: 70,000 km ÷ 10,000 km per change × 27.6 Litres = 193.2 Litres per year

Used lubricating oils in most jurisdictions are classified as hazardous waste, as is the case in Manitoba (Government of Manitoba 2004), and, as such, there are strict requirements for handling and disposal. At the same time, the use of lubrication oils is very widespread, and the potential damage that can be caused from improper disposal is significant (Madanhire and Mbohwa, 2016). As such, collaboration between provinces in Canada began in 1993, under the auspices of the Canadian Council of Ministers of Environment (CCME), to create industry-wide programs to address waste oils, based on a stewardship model. The result has been a series of provincial-based industry programs across the country, coordinated through the Used Oil Management Associations of Canada (UOMAC).

The responsible organization for waste oil management within Manitoba is the Manitoba Association for Resource Recovery Corporation (MARRC), a non-profit created in 1997. MAARC is managed by a board of directors, which is comprised of stakeholders. The mandate is to implement an economical, simple, accessible, and sustainable waste oil management program on behalf of the industry members.

A key aspect of the programs across Canada is the collection of an Environmental Handling Charge (EHC) on all first-use products. Within Manitoba, this charge is currently $0.10 per Litre for lubrication oil (UOMAC 2010). This charge is part of the cost of input materials to periodic oil changes, and is thus part of regular maintenance costs, and is not included for analysis here. It is this charge that funds the activities of MAARC and other similar organizations.

For individual consumers and small oil users, there are series of depots available where waste oil can be dropped-off at no charge. In these cases, MAARC provides a reimbursement to the collector involved. This value ranges from $0.08 to $0.18 per Litre for used-oil, depending on location. Large oil users need to work through a licensed collector to have their used-oil collected. However, this business market is extremely competitive, with specific charges not publicly released. For the purpose of analysis, a mid-range reimbursement value from MAARC is used as an approximate surrogate. A value of $0.12 per Litre is assumed, this to also cover off other small amounts of waste fluids that may also be involved, such as used anti-freeze solutions.

Based on these assumptions, the estimates for present value costs of used-oil disposal are provided as follows:

- **Lower annual travel**: 96.6 Litres per year × $0.12 per Litre × 9.20 PVIFA = $110 present value
- **Average annual travel**: 138.0 Litres per year × $0.12 per Litre × 9.20 PVIFA = $150 present value

- **Higher annual travel**: 193.2 Litres per year × $0.12 per Litre × 9.20 PVIFA = $220 present value

These values are relatively small. Importantly, although waste oil could pose a serious environmental concern, this issue now appears to be suitably addressed.

14. Acid Precipitation Emissions

The reduction of acid precipitation emissions provides a positive contribution with the transition from diesel to electric transit buses, however, the monetized values involved are not consequential. The results suggest that the impacts of this once significant environmental problem have been largely addressed, through implementation of ultra-low sulphur diesel, and that transitioning to electric buses today provides little incremental benefit in this regard.

The phenomenon of acid precipitation has been known for over two hundred years. It was only into the 1970s, however, that it was recognized as a serious environmental threat. Across Eastern North America and Northern Europe, noticeable declines in lake health were identified, particularly in relation to the stability of fish populations in lakes near heavily industrialized areas. Pioneering work at the Experimental Lakes Area (ELA) in Northwestern Ontario helped confirm the empirical link between industrial pollution and the observed decline in lake and fish health (e.g., refer to Schindler 2009). Specifically, research helped establish that coal-fired electricity production was a dominant source of sulphur dioxides, with these emissions in turn contributing to the acidification of water bodies.

Acid precipitation, more popularly known as “acid-rain,” can involve dry deposition, but occurs primarily due to the progressive atmospheric reaction of excessive gaseous oxide emissions to ultimately form water-soluble acids that subsequently fall with rain (Bailey et al. 1978). The primary emissions of concern are sulphur oxides (SOx), in particular sulphur dioxide (SO2). The latter is normally present in the atmosphere but at very low levels, around 0.2×10^{-6} % by volume. Sulphur oxides can be ultimately converted in the atmosphere to sulphuric acid (H2SO4), which is a strong acid, contributing to roughly 70% of the problem. Nitrogen oxides (NOx) also can play a role, given some of these can be ultimately converted to nitric acid (HNO3), again a strong acid. These contribute to roughly 30% of the problem. Carbon dioxide (CO2) too can be converted to carbonic acid (HCO3), however this is a weak acid and much less consequential. Sulphur oxides and nitrogen oxides are mostly not GHGs, so they are today often overlooked when analyzing the environmental impacts of a process. Nevertheless in the late 1970s and early 1980s, acid precipitation was recognized literally as the most serious immediate environmental problem facing humankind (Elsworth 1984).

Since the late 1970s, a variety of international agreements have come into effect to ensure consistency for reduction efforts on sulphur oxides. Emissions of sulphur oxides have been steadily declining, representing within Canada alone roughly a three fold reduction since 1990, i.e., 3.1 million tonnes per year in 1990 down to 1.1 million tonnes per year in 2014 (ECCC

2016). A major part of this relates to coal-powered electricity production, including both improved emission controls and a general transition away from coal altogether. Increased awareness and other specific programs have also contributed. The connection to transit buses is through the sulphur content of diesel fuel. Up until 2006, a relatively high level of sulphur had been permitted in diesel, representing upwards of 500 parts per million by mass. The coordinated introduction of ultra-low sulphur diesel (ULSD) in Canada, the U.S. and Europe began in 2006, requiring a much lower mandated elemental sulphur content of no more than 15 parts per million by mass (i.e., 97% reduction). The legal requirement for this in Canada is the Sulphur in Diesel Fuel Regulations, under the Canadian Environmental Protection Act. Given the higher molecular weight of a sulphur dioxide molecule (i.e., 64) compared to sulphur itself (i.e. 32), consumed-fuel sulphur content multiplied by 2 provides an estimate of the resulting sulphur dioxide emissions.

Based on the assumed diesel fuel consumption, including biofuel mandate, and travel distances, as noted previously, net higher annual sulphur dioxide emissions associated with diesel buses compared to electric (i.e., taking account of small amount of diesel used for auxiliary heating) are summarized as follows:

- **Lower annual travel**: 0.52 kg SO2 per year net for diesel bus
- **Average annual travel**: 0.74 kg SO2 per year net for diesel bus
- **Higher annual travel**: 1.04 kg SO2 per year net for diesel bus

Although these are significant reductions for electric buses in relative terms, the values, in absolute terms, are all very small.

A surrogate value for damages from sulphur dioxide emissions has been available since 1990 in the U.S. based on allowances for SO2 emissions from electric utilities that have been capped and traded through an auction market. A report by researchers at MIT (Joskow et al. 1996) indicated that a private market for trading allowance permits had initially emerged, but current market statistics indicate this private market has since collapsed (Cooper et al. 2012).

As outlined by the EPA (2017), the 2017 allowance spot auction had no privately offered allowances and only 125,000 total (all from the EPA reserve). Spot bids ranged from $0.04 to a high of $0.75 for a total of 125,000 tons worth of allowances. Allowances can be carried forward and used in subsequent years, therefore the low demand for allowance permits possibly suggests market saturation or the redundancy of the program due to lowered emission levels.

There had been substantial volatility in the allowance auction market, at least during its the first two decades of operation. The weighted average price for winning bids in the 1996 spot auction was $68.05 with the highest bid at $300.00. In 2006, Spot prices had jumped to a weighted average of $883.10 and a highest bid of $1,700.00. This price volatility indicated unpredictability in demand for emission allowances, likely related to the grid-mix.

Decreased volatility in the allowance future market today indicates greater faith in market stability. Seven-year advance auction (not useable before 2024) sold 125,000 of 250,100 bids with prices ranging from $0.25 and a clearing price of $0.01. Weighted average of winnings bids was $0.01. The demand for future emission allowances may indicate that the market believes emission levels to be stable and at the likely minimum level for the foreseeable futures grid mix.

For analysis purposes, this highest 2017 bid price has been employed, namely about $0.75 USD per ton SO2. This translates to approximately $1.04 per tonne for annual emission reductions here. Combined with a PVIFA value of 9.20, and dividing by 0.70 to account for other acid emissions as noted earlier, provides the following total monetized present value impact results:

- **Lower annual travel**: 0.52 kg SO2 × $1.04/tonne × 9.20 PVIFA ÷ 0.70 = $0.007 present value – this value is negligible
- **Average annual travel**: 0.74 kg SO2 × $1.04/tonne × 9.20 PVIFA ÷ 0.70 = $0.010 present value – this value is negligible
- **Higher annual travel**: 1.04 kg SO2 × $1.04/tonne × 9.20 PVIFA ÷ 0.70 = $0.014 present value – this value is negligible

These resulting values obviously are negligible. However, the SO2 reductions achieved through the implementation of ULSD are still important to acknowledge. For Canada alone, total diesel fuel consumption for 2013, after ULSD implementation had been largely completed, was about 30 billion Litres (Fuels Canada Association 2014), i.e., 28% of total domestic sales of refined products of 107 billion Litres. Using that volume of fuel, the reduction of SO2 emissions comparing pre-2006 diesel to ULSD levels translates to approximately 24,000 tonnes per year.

15. Used Battery Final Disposal

A last component issue examined is the impact associated with final disposal of lithium-ion batteries, as employed in electric buses, at the end of their useful life. Based on the principle of supplier responsibility, it is assumed that battery- and/or bus-manufacturers will be likely assigned responsibility to address used-batteries as part of any procurement agreements. This component issue is thus unlikely to result in any appreciable externalities, i.e., costs will be internalized as part of purchase prices.

The analysis undertaken, however, reveals a troubling need within Canada, and indeed the rest of the world as a whole, to be prepared both economically and physically for the disposal of large numbers of such batteries. Battery-based buses, and electric vehicles in general, are simply too new, i.e., no more than roughly six to eight years old at most, such that almost no batteries have yet reached the end of their intended life. It is anticipated that no significant quantities of used lithium-ion batteries from vehicles will require disposal for many years, yet in time disposal quantities could rapidly escalate. For example, forecasting by Standridge and

Corneal (2014) suggests by 2035, the number of vehicle-related lithium-ion batteries reaching end-of-life could range from 1.4 million to 6.8 million in the U.S. alone.

Lithium-ion battery packs are considered dangerous goods for transportation within Canada and elsewhere (Transport Canada 2016), with the primary restrictions related to air-transport. This approach has been more cautionary in nature given a number of incidents that have occurred on-board aircraft. Lithium batteries themselves are not inherently hazardous, but can become a concern under certain specific circumstances. As noted by Huo et al. (2017), they can become thermally and electrically unstable if subjected to certain uncontrolled environmental conditions or mishandled during transportation. Additionally, some lithium-ion battery chemistries are intrinsically safer that others by design, e.g. LTO and LFP being noted for enhanced safety characteristics (Recharge 2013). As such, in terms of hazard, lithium-ion batteries have been grouped as a whole under Class 9, i.e., miscellaneous hazardous materials. Transport restrictions also only apply depending on the volume and capacity of batteries involved, being much less restrictive for small quantities of low-powered batteries (i.e., less than 100 Wh).

This is very different from the situation with conventional lead-acid batteries. These may vary somewhat in physical format and arrangement but all possess essentially the same chemistry. Aside from being electrically charged, the two main components of these are individually hazardous on their own, i.e., lead, which is toxic, and concentrated sulphuric acid electrolyte, which is highly corrosive. Overall for lead-acid batteries the dominant hazard in terms of dangerous goods is their corrosive nature, such that they are grouped under Class 8, i.e., corrosive materials.

Simply disposing old lithium-ion batteries by dumping them in landfills would be highly wasteful, and improper disposal also could lead to potential for groundwater pollution (Taylor 2009). Applying some sort of reuse, recovery or recycling approach thus makes most sense. As outlined by CM Consulting (2012), there are two main processes that can be included as part of in recycling and recovery operations:

- Hydrometallurgical processes, which use water and chemicals to recover constituents and metals; and
- Pyrometallurgical processes, which break down batteries through thermal treatment.

Into the future three basic options have been outlined for dealing with lithium-ion battery end-of-life, in terms of recycling or recovery (Gaines 2014):

- **Smelting** involves high-temperature processing that consumes organic constituents and reduces the battery components down to residual metals, including cobalt, copper, nickel, iron and others. This is the simplest easiest approach for recycling, being entirely pyrometallurgical in nature, but does not allow the batteries to remain directly part of the supply chain. It is also energy intensity and requires the flexibility of accepting a diverse array of battery chemistries. The positive side is that smelting does recover potentially scarce metals, in particular cobalt as noted earlier (see page 16), can be valuable for resale and can be readily reused again to manufacture new batteries.

- **Intermediate recycling** involves the separating of plastics and metals, typically via hydrometallurical methods, and then uses liquid filtration to separate metals such as aluminum to reclaim and recycle. It is more time consuming and costly, and is likely only economical if a recycler can guarantee to reclaim the more valuable cobalt and nickel components from the battery. Such processes show promising results, but have yet to be employed at large scale.
- **Direct recycling** recovers most of the battery materials for reinsertion into the battery supply chain. All the materials are recovered, including the valuable cathode constituents.

A further contrast with conventional lead-acid batteries is important to note. Lead-acid batteries involve relatively standardized components, fewer components in total, and less complex construction. It is thus not surprising that lead-acid batteries have the highest recycling rate of any consumer product in the world, in the range of 95% to 100% (Gaines 2014). Lithium-ion batteries, on the other hand, are more complex in construction, much more diverse in terms of composition, and more composite in nature. As such, they are much more difficult to recycle or reuse.

Taylor (2009) noted that eight years ago, there was only one significant company within the U.S. involved with the recycling of lithium-ion batteries, this being California-based Toxco Inc. They were renamed Retriev Technologies in 2013, and remain one of only a very small number of companies identified in this nascent business area, including one of their bases of operations in Trail, B.C. Their focus has been hydrometallurgical processing. CM Consulting (2012), at that time, noted only two recycling companies with processes for dealing for lithium-ion batteries in Canada: Retriev Technologies (Toxco), as noted above; and Glencore (formerly Xstrada), noted below. Glencore is a large international mining and mineral-processing firm. Included in its assets is the Sudbury Integrated Nickel Operations smelter in Sudbury, Ontario (Glencore ND). This operation includes used-batteries, including lithium-ion batteries, as part of its "custom feeds" input. This operation was formerly under Xstrada, and prior to that was known under the name Falconbridge. Other major international processors include Belgium-based Umicore, and Japan-based JX Nippon Mining & Metals Corporation (2012). There are also a number of smaller development-stage companies. Although there are a number of hydrometallurgical processors, the primary orientation in recycling is still primarily on smelting and pyrometallurgical methods.

As described earlier in the section on scarce minerals (see page 16), the most valuable constituent of lithium-ion batteries is typically the cobalt, which is part of the cathode, and can be readily recovered via smelting. The common six battery chemistries can be ranked approximately as follows in terms of relative content of cobalt (Recharge 2013):

LCO > NMC > NCA > LTO > LMO or LFP

In terms of vehicle applications, NMC is most relevant for electric buses. This has the second highest content of cobalt compared to other common chemistries. LCO is generally not used in vehicular applications, instead being mostly employed for consumer electronics, e.g. cellular telephone, laptop computers, tablet computers, video cameras, etc. These batteries are generally much smaller, but contain a significantly higher proportion of cobalt (i.e., roughly 4 times higher than NMC), which makes them inherently more valuable for cobalt recovery. NCA, which is also used extensively for vehicle applications, has only roughly half the cobalt content of NMC. This makes recovery and recycling for NMC to be somewhat trickier if looking just at cobalt. The cobalt content is enough to be a concern in terms of scarce mineral costs (see page 16), but much less lucrative for cobalt recovery than LCO. Using more of an intermediate or direct recycling approach may be needed to achieve adequate economics.

A final, fairly obvious option to consider for vehicle-related lithium-ion batteries is reuse, in particular secondary purposing for stationary electrical applications as energy storage systems or ESS (e.g., refer to Walker et al. 2015). In the past this approach held out much promise, significantly extending the potential life of batteries and gaining more useful value. However, on closer examination, more recent evaluations have found such an approach to be not so attractive (e.g., Lux Research 2016 or Robinson 2017). There are multiple concerns, including the logistics and costs of collecting sufficient batteries from disparate sources, the uncertain and uneven status of batteries and associated battery-balancing issues, and strong competition from advanced, purpose-build batteries for ESS applications.

It is clear that battery end-of-life considerations for electric buses and other types of electric vehicles remain unclear, simply because there has been little need so far to address this in a meaningful manner. This situation presents not just a potential headache, but also a significant possible opportunity. Much more work is certainly required.

16. Combined Results

Off the twelve component issues considered, nine provide a contribution that is positive for the transition to electric buses, whether or not each may be significant. Three on the other hand provide a contribution that is negative. The positive component issues, in order of contribution, are: greenhouse gas emissions; diesel fuel price volatility[*]; noise impacts; photochemical smog precursor emissions; particulate matter emissions; carbon monoxide emissions; air toxics emissions; used diesel engine lubricant oil disposal; and acid precipitation emissions. The negative component issues, in order of contribution, are: rare battery mineral scarcity; weight-induced infrastructure damage; and used battery final disposal.

[*] As noted in the section starting on page 7, if evaluated on a purely theoretical basis, diesel fuel price volatility does produce the second highest monetized value, but in such a way that likely would not be realistically undertaken, and as such a reduced value is included that is more practical in nature.

The monetized present value results for all component issues are summarized in **Table 3**, with positive contributions in **black** and negative contributions in **red**. The net overall present value of combining all contributions for the issues is positive for electric buses compared to conventional diesel buses. The net value depends on assumed annual travel, ranging from about $33,000 overall for lower assumed annual travel to about $74,000 overall for higher assumed annual travel. Indeed, the net overall result depends more strongly on annual travel than for any individual component issue. This result emphasizes the need in terms of economic viability to have electric buses travel as much as realistically possible.

Table 3. Summary of Monetized Values for All Externality Factors Investigated

Component Issue	**Present Value of Impact on Electric Bus**		
	Lower Travel	**Average Travel**	**Higher Travel**
1. Greenhouse Gas Emissions	$21,250	$30,360	$42,500
2. Diesel Fuel Price Volatility	$10,660	$15,230	$21,320
3. Noise Impacts	$11,990	$17,280	$23,980
4. Rare Battery Mineral Scarcity	($8,300)	($8,300)	($8,300)
5. Weight-Induced Infrastructure Damage	($5,520)	($7,880)	($11,040)
6. Photochemical Smog Precursor Emissions	$1,970	$2,810	$3,940
7. Particulate Matter Emissions	$480	$690	$960
8. Carbon Monoxide Emissions	$160	$220	$320
9. Air Toxics Emissions	$130	$180	$260
10. Used Diesel Engine Lubrication Oil Disposal	$110	$150	$220
11. Acid Precipitation Emissions	Negligible	Negligible	Negligible
12. Used Battery Final Disposal	Negligible	Negligible	Negligible
Overall Net Total	$32,930	$50,740	$74,160

In terms of relative contributions, three of the twelve component issues are relatively high, i.e., present values in the tens of thousands of dollars, and are thus most significant. These are: greenhouse gas emissions; diesel fuel price volatility; and noise impacts. Three of the component issues provide moderate contributions, i.e., present values in thousands of dollars. These are: rare battery mineral scarcity; weight-induced infrastructure damage; and photochemical smog precursor emissions. All of the other component issues provide only small contributions, with present values all less than $1,000, and are, as such, much less important overall.

Many of the component issues are related. Results can be recalculated, grouping issues into five relevant clusters. These are presented in **Table 4**. Importantly, this presentation does not alter the results as described earlier, with the top four component issues in order still dominant.

Table 4. Summary of Monetized Values Clustered into Related Component Groups			
Clustered Component Issues (Issues Included Together)	**Present Value of Impact on Electric Bus**		
	Lower Travel	**Average Travel**	**Higher Travel**
Greenhouse Gas Emissions Related (#1)	$21,250	$30,360	$42,500
Noise Related (#3)	$11,990	$17,280	$23,980
Volatility and Scarcity Related (#2 and #4)	$2,360	$6,930	$13,020
Infrastructure Damage Related (#5)	($5,520)	($7,880)	($11,040)
Air-Pollution Related (#6, #7, #8, #9 and #11)	$2,850	$4,050	$5,700
Used Material Disposal Related (#9 and #12)	$110	$150	$220
Overall Net Total	$32,930	$50,740	$74,160

Several additional important results emerge from the analyses, summarized as follows:

- Noise-related impacts for electric buses are, perhaps surprisingly, much more significant than air pollution-related component issues, given the relatively noisy nature of diesel vehicles and the very quiet nature of electric buses. Yet, electric buses can be considered as too quiet, and adequate acoustic alert systems may be required to protect pedestrians.

- Nitrogen oxides represent the most important of the air pollution-related component issues considered, at least in terms of Winnipeg and Manitoba. Health Canada (2016) noted that nitrogen oxide levels in Manitoba, particularly within Winnipeg, are relatively high compared to other parts of the nation as a whole.

- Two component issues, used lubrication oil disposal and acid precipitation emissions, are known in the past to have been highly important, but today have been largely addressed, through national used-oil stewardship programs and ultra-low sulphur diesel fuel respectively. Both of these issues are now near the bottom in terms of relative impacts, and monetized value.

- Two important cautionary issues are identified that will need to be considered more closely into the future: corporate social responsibility associated with rare battery minerals extraction; and processes for recovery and recycling of battery components in terms of final battery disposal.

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