CASE STUDY

Interpretation of 24-hour sampling data: Development of 24-hour ambient air quality criteria and their use in Ontario

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1.0   Introduction

The Ontario Ministry of the Environment (MOE) sets science-based ambient air quality criteria, or AAQCs, to evaluate regional air quality data. An AAQC is a desirable concentration of a contaminant in air that is unlikely to adversely affect human health or the environment. The term "ambient" is used to reflect general air quality independent of location or source of a contaminant. AAQCs are most commonly used in environmental assessments, special studies using ambient air monitoring data, assessment of general air quality in a community and annual reporting on air quality across the province

Ontario's 24-hour AAQCs are based on health effects and are set at concentrations that are protective against effects that may occur during continuous lifetime exposure. In comparison, the Texas Commission on Environmental Quality develops reference values to be used as 24-hour Air Monitoring Comparison Values (AMCVs), to compare to measured 24-hour ambient air concentrations although the TCEQ also develops acute 1-hr and chronic AMCVs to evaluate 1-hr measured concentrations of chemicals or calculated annual average concentrations, respectively. This case study describes the Ontario approach and discusses how the Ontario AAQCs and Texas ACMVs may be applicable, depending on the science and implementation considerations.

1.1   Assigning Averaging Times

Following conversions to a continuous exposure in generating an AAQC, , an averaging time is assigned. It represents the time period over which air quality data are collected for comparison to the AAQC. In general, averaging time selection may be influenced by both the underlying toxicology of a substance, including exposure and effects (largely governed by science judgment), and implementation considerations, including modelling and monitoring (largely governed by science policy).

The Ontario Ministry of the Environment (MOE) currently takes into consideration the duration of exposure associated with effects, nature of the relevant critical effect(s), and mode of action when assigning an averaging time. For AAQCs that are protective against effects that may occur during continuous lifetime exposure, an annual averaging time is assigned when effects are caused over longer-term exposures and a 24-hour

averaging time is assigned when effects are caused after short-term exposures (e.g. developmental) and/or when daily variation remains a concern.

A challenge of an annual AAQC, however, is that air quality can only be assessed after sufficient air quality data are collected to reflect an annual average. That is, annual AAQCs are not useful for evaluating individual 24-hr exposures. The MOE has addressed this issue by converting AAQCs with annual averaging times to 24-hr AAQCs via a meteorological-based conversion factor. In that case, two AAQCs are set for a single effect: an effects-based annual average AAQC, and a converted 24-hr AAQC.

Figure 1 illustrates how averaging times are set. An AAQC may be assigned an averaging time of 24 hours based on: 1) concerns about effects that may develop after short-term exposure periods; or 2) a conversion from the annual averaging time to facilitate more timely assessment of air quality. These two approaches for setting 24-hour AAQCs are described in further detail below. It is important to keep in mind when reviewing these approaches that the 24-hour AAQCs set by the MOE are protective against effects from continuous lifetime exposures to a contaminant.
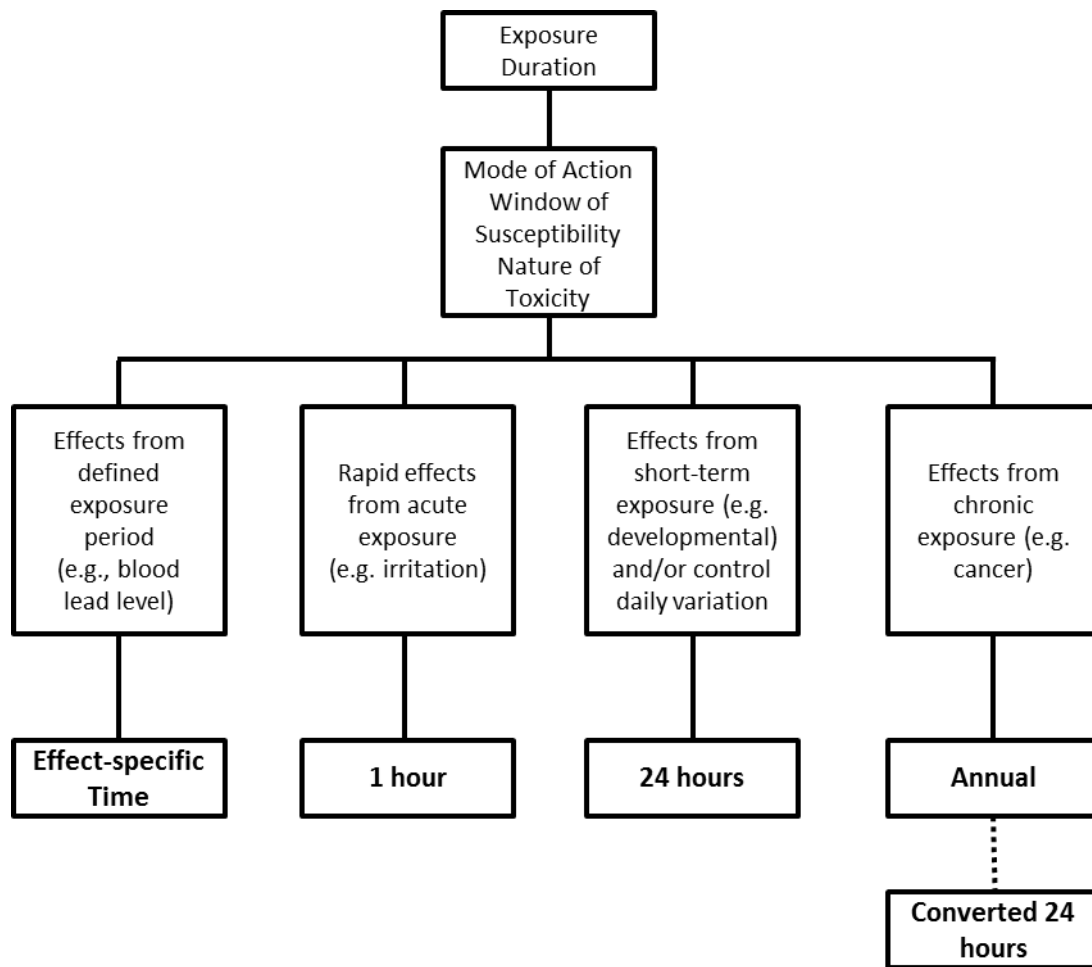

Figure 1: MOE Approach for Selection of an Averaging Time

1.2   Application of AAQCs

AAQCs are used to assess monitored air quality resulting from the contributions of a contaminant to air from all sources. AAQCs are also used to set regulatory air standards in Ontario, which are usually the same numerical value as AAQCs but are different in how they are used.   Air standards are used to evaluate the contributions of a contaminant to air from a single regulated source, most often through modelling but sometimes through a combination of modelling and monitoring. Air standards are used to assess regulatory compliance, identify needs for abatement and also to inform permitting decisions. While the focus of this case study is on the AAQC component of our air quality program, it is also relevant to air standards.

2.0   Establishing AAQCs for Continuous Lifetime Exposure

In establishing an AAQC, the MOE reviews approaches taken by agencies in Canada and around the world (i.e., Health Canada, World Health Organization (WHO), International Agency for Research on Cancer (IARC), United States Environmental Protection Agency (US EPA), European Union, United Kingdom, Texas and several relevant U.S. state agencies) and reviews the recent scientific information related to the toxicology of the substance, including effects on human health and the environment. From this, the dose-response relationships for an array of adverse effects considered critical are assembled.   Once a point of departure is designated from the most appropriate adverse effect, an application of uncertainty factors (for threshold effects) and/or extrapolation (for non-threshold effects) occurs and a concentration in air is calculated based on an assumed continuous lifetime exposure.   Where appropriate, media allocation is taken into consideration. The final step in the AAQC setting process is to assign an averaging time.

2.1   24-hour AAQCs: Approach 1 – Effects caused after short-term exposure

The MOE assigns an averaging time of 24 hours to AAQCs based on effects caused by short-term exposures (e.g., reproduction, development). A recent example of this approach was the development of an AAQC for dioxins, furans, and dioxin-like PCBs, where a 24-hour averaging time was selected due to a developmental endpoint with fetal exposure uncertainties.

Example: Dioxins, Furans, and Dioxin-Like PCBs (24-hour averaging time)

A weight-of-evidence approach was used in considering which path to take in establishing an AAQC for dioxins, furans, and dioxin-like PCBs (MOE, 2011a). The approach proposed by the WHO is used by many regulatory agencies around the world (with the exception of the US EPA) and was considered to be the most appropriate.

These agencies focused on the critical effects observed at the lowest body burdens as a basis for setting health protective values for dioxins and dioxin-like compounds. They calculated the tolerable intake values by dividing the NOAELs/LOAELs for these effects

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by a number of uncertainty factors intended to take into account several sources of uncertainty. In rodent models, the developmental effects (namely, the effects on male reproductive tract growth and development) in the offspring of dams exposed to TCDD during gestation appeared to be the most sensitive endpoints of TCDD toxicity. Thus, they were considered to be the critical effects on which the development of an AAQC for dioxins, furans, and dioxin-like PCBs should be based.

The study by Faqi et al. (1998) was selected as a key study. The LOAEL identified in this study resulted in animal steady-state maternal body burden of 33 ng/kg bw. Assuming TCDD half-life of 2774 days and 80% bioavailability from food, the equivalent human daily intake (EHDI) of 10.3 pg/kg bw/day was calculated. Application of the composite uncertainty factor of 10 resulted in the toxicity reference value (TRV) of 1.0 pg/kg bw/day. Application of a 3% apportionment of the TRV to exposure from air and subsequent route-to-route extrapolation (assuming an inhalation rate of 20 m^3/day and a body weight of 70 kg for an average person) resulted in an AAQC value of 0.1 pg/m^3 (rounded figure). 100% absorption was assumed given the fact that dioxins are found in air on fine airborne particulate and also in vapour phase.

An AAQC for dioxins, furans, and dioxin-like PCBs is applied to 29 specific compounds of concern that exhibit biological activity similar to that of 2,3,7,8-tetrachlorodibenzo-p-dioxin, the most potent compound within these groups of chemicals: 7 dioxins, 10 furans, and 12 dioxin-like PCBs. In terms of application, a speciated air mixture of PCDDs/PCDFs/dioxin-like PCBs should first be converted to toxic equivalents by multiplying the concentration of each congener by the appropriate WHO_{2005} toxic equivalent (TEQ). Summing the individual TEQ values for each congener would provide the total toxicity equivalent level for the air mixture. An AAQC for dioxins, furans, and dioxin-like PCBs is expressed in units of WHO_{2005}-TEQ per cubic metre and applied to the total emission of dioxin-like compounds (i.e., particulate and vapour-phase emission).

Considering the existence of the critical windows of vulnerability for the effects on developing male reproductive organs and also the uncertainty in estimating fetal exposure, the MOE selected a 24-hour averaging time.

Thus, a 24-hour average AAQC of 0.0000001 µg WHO_{2005}-TEQ/m^3 for dioxins, furans, and dioxin-Like PCBs, based on the developmental effects associated with exposure to these compounds, was established. If the concentration of these compounds does not exceed the AAQC, no adverse effects are expected in continuous lifetime exposures.

Limitations of the Approach: Effects-based 24 hr AAQC

The MOE's effects-based 24-hour AAQC can be used to set targets for air quality and can be used to readily assess air quality relative to these targets. If the AAQC is met then no adverse effects are expected over continuous lifetime exposure. Since an AAQC protective of lifetime exposure is generally more stringent than one protective of higher dose short-term / acute exposures, this AAQC is also likely protective against rapid effects from short exposures. However, this AAQC is not appropriate for

assessing the health risk associated with single or periodic 24-hour exposures, since the numerical value of the 24-hour AAQC is much more stringent than appropriate to evaluate single or periodic 24-hour exposure scenarios. For such exposure scenarios, Ontario relies on case-specific scientific analyses and/or other regulatory tools.

2.2   24-hr AAQCs: Approach 2 - Conversion from Annual AAQCs

Similar to what is done by the TCEQ, the New York State Department of Environmental Conservation (NYSDEC), and California Office of Environmental Health Hazard Assessment (OEHHA), the MOE assigns annual averaging times to AAQCs for contaminants causing adverse effects after chronic exposures.

From the toxicological perspective, the annual AAQC is protective against effects that may be expected from continuous lifetime exposures. However, the annual AAQC does not allow assessment of short-term periods of elevated exposure that may cause a different effect from that used to set the annual AAQC, or increase the risk of the same effect used to set the annual AAQC. Another limitation of the annual AAQC is that air quality can only be assessed after sufficient air quality data are collected to reflect an annual average. That is, longer averaging times require more sampling and longer delays in order to get enough data to compare to an air quality criterion.

To address some of the limitations of the utility of the annual AAQC, the MOE converts the annual AAQC to a 24-hour value using a meteorologically-based conversion factor and the *converted* 24-hour AAQC is used to assess 24-hour air quality data (See Appendix A). If the converted 24-hour AAQC is met, it is assumed that the annual AAQC will not be exceeded and no effects are expected over continuous lifetime exposure. Theoretically, the converted 24-hour AAQC is also protective against potential effects of higher-dose short-term exposures (as long as the conversion does not result in a value that is above a concentration of concern for another adverse effect). The development of annual AAQCs and converted 24-hour AAQCs is described below.

Development of Annual AAQCs

The rationale underlying the development of three recent annual average AAQCs by MOE is presented below: benzene, 1,3-butadiene and uranium.

Example: Benzene AAQC (annual averaging time)

Numerous findings from *in vivo* and *in vitro* studies indicate that benzene is carcinogenic to both humans and animals. While several types of leukemia have been observed in epidemiological studies, acute myeloid leukemia is the subtype of leukemia that has generally been used by agencies for quantitative dose-response analysis.

Recent publications on modes-of-action suggest that both genotoxic and epigenetic mechanisms may be responsible for the toxicological events leading to carcinogenicity at low doses. Weight-of-evidence from recent toxicokinetic and toxicodynamic studies indicate a genotoxic component to benzene carcinogenicity at low doses. Certain observations, on their own, *qualitatively* provide support that either a sublinear,

supralinear, or linear response may occur; however the data remains insufficient for *quantitative* dose-response analyses, based on sub- or supralinearity. Low dose linear extrapolation from high doses using linear models is a common approach for compounds with a genotoxic mode-of-action.

Considering the overwhelming evidence of the strong carcinogenic potential of benzene, regulatory agencies have established air guidelines towards reducing potential cancer risks. Because the carcinogenic effects of benzene may occur at lower levels of exposure than the non-carcinogenic effects, basing guidelines on the mitigation of potential cancer risk also provides the necessary protection against the development of non-carcinogenic effects.

Many regulatory agencies have used the Pliofilm cohort as the best published data set to quantitatively evaluate human cancer risks from exposure to benzene. The methods of the US EPA (linear) and the EU (linear and non-linear) are considered to be the most appropriate approaches, as they employ multiple models for dosimetric analyses of the Pliofilm data. In both cases, the recommended air quality criteria were derived from extrapolation of occupational exposure concentrations to ambient air exposures. Additionally, there are some key toxicokinetic and toxicodynamic issues specific to benzene that have been considered in various studies, when extrapolating from high to low dose (i.e., studies published subsequent to the US EPA and EU analyses). These more recent studies have begun to shed light on the toxicological events that may occur at environmental exposure concentrations – namely, DNA repair kinetics, adduct formation, and the benzene metabolism pathway at relevant low doses. Overall, the molecular toxicological events provide a weight-of-evidence support for a health protective linear extrapolation method for the development of benzene AAQC. Such an approach has been performed by the USEPA, though the USEPA performed linear extrapolation as a default approach.

Ontario's practice is to set AAQCs for carcinogens at concentrations equivalent to an incremental lifetime cancer risk of 1 in a million (or 10^{-6}). Based on an evaluation of the scientific rationale of air criteria from leading agencies, an examination of current toxicological research, and comments from stakeholders, the following effects-based AAQC was set for benzene, adopting the inhalation unit risk of the US EPA with further toxicological reasoning for linear extrapolation: an annual AAQC of 0.45 µg/m^3 (0.14 ppb) for benzene, based on carcinogenicity associated with exposure to this compound, was developed (MOE, 2011b).

Example: 1,3-Butadiene AAQC (annual averaging time)

Symptoms of chronic human exposure to 1,3-butadiene in air may include a variety of cardiovascular and haematological effects, as well as nasal cavity irritation. It has been observed that the metabolites of 1,3-butadiene can also induce mutational changes in proto-oncogenes and tumour suppression genes, as well as introduce clastogenic damages to the genetic materials in mice, rats, and humans, via both in vitro and in vivo studies.

In animals, increased incidences of and death from tumours in various tissues from exposure to 1,3-butadiene have been reported. In humans, increased mortality from lymphosarcoma and leukemia have been observed at 1,3-butadiene monomer and styrene-butadiene rubber manufacturing facilities, respectively. The reactive metabolites of 1,3-butadiene are capable of reacting with genetic materials that may result in forming covalent adducts with DNA. All the agencies and jurisdictions reviewed have derived air quality criteria based on the carcinogenic potential of 1,3-butadiene.

Epidemiological data from cohorts of 1,3-butadiene monomer manufacturing facilities did not provide a reliable basis for cancer risk assessments because of the small size of the studies, and the inconsistent significance in the findings of the identified lymphatic cancers. However, a detailed risk analysis of workers exposed to 1,3-butadiene at eight styrene-butadiene rubber manufacturing facilities across North America demonstrated a significant increase in mortality rates of leukaemia. Four jurisdictions in the US and Canada have derived their respective inhalation unit risk estimates based on the data of this analysis: Environment Canada/Health Canada, the US EPA, the states of Texas and North Carolina.

Considering that the exposure estimates of this analysis may be under-estimated, leading to over-estimated cancer risk estimates, the states of North Carolina and Texas have incorporated updated exposure data from follow-up studies. Thus, among the four sets of inhalation unit risk factors developed by these jurisdictions, the cancer risk estimates developed by the states of Texas and North Carolina are much improved unit risk values as compared with those developed by Environment Canada/Health Canada and the USEPA, considering their use of updated and more reliable exposure estimates in the dose-response analyses. Of these two, the unit risk estimate of the State of Texas (TCEQ) is considered to be the most appropriate as its derivation is based on an extensive risk analysis and the involvement of a detailed peer review process.

Adopting inhalation unit risk of the TCEQ, an annual average AAQC of 2 µg/m^3 (0.9 ppb) for 1,3-butadiene, based on carcinogenicity associated with exposure to this compound was established (MOE, 2011c), which reflects an incremental lifetime cancer risk of 10^{-6}.

Example: Uranium non-cancer AAQC (annual averaging time)

This AAQC development provides to be an interesting example, as the MOE considered the fact that uranium can be toxic to humans due to its chemical (i.e. renal damage) and radiological properties (i.e. carcinogenic potential). Briefly, the direct effects of uranium on the kidney are well established from both human and animal studies and, as such, have been recognized by health organizations and regulatory agencies to be the appropriate regulatory endpoint to develop health protective limits. In contrast, there is insufficient toxicological evidence for an increased risk of cancer associated with uranium exposure. Together with additional considerations, the MOE has determined that kidney toxicity is the most sensitive endpoint associated with exposure to uranium and uranium compounds, on which to base the AAQC. An annual averaging time was

assigned to this contaminant in recognition that the effect on the kidney was modelled over a 50-year time period.

The MOE has relied on human biokinetic models as a methodology for developing the uranium AAQC, gleaning knowledge from modeling the kidney accumulation of uranium, together with the reported information from studies done in experimental animal models and the findings from epidemiological studies assessing the kidney toxicity of uranium, as summarized below:

- The toxicity of various uranium compounds varies depending on their physico-chemical properties, with solubility as one of the most relevant characteristics that may influence their toxicity. The International Commission of Radiological Protection distinguished three classes of uranium compounds based on their solubility in bodily fluids: fast (F),medium (M) and slow (S), reflecting the rate of absorption of these compounds into the body that can be estimated from the biokinetic model. From a kidney perspective, those uranium compounds that are the most soluble tend to be the more toxic.
- A reference concentration of 0.1 µg U/g kidney as recommended by the United Kingdom's Royal Society is preferred as an appropriate assessment to be used as a point-of-departure for the AAQC. The origin of the recommendation is a modelled estimate of a uranium kidney concentration from an epidemiological study that investigated the effects of chronic uranium exposure from drinking water on subtle adverse kidney effects.
- The ICRP-based human biokinetic models were considered appropriate for developing an air criterion, and allowed for several considerations presented above to be simultaneously integrated in developing air criteria. These include:
  - Age-specific physiological and uptake parameters.
  - Allocation of 10% to inhalation exposure to uranium, limiting the kidney burden to 0.01 µg U /g kidney over 50 years. An allocation of 10% to limit the kidney burden attributable to inhalation exposure takes into consideration other routes of uranium exposure that can contribute to kidney effects.
  - Absorption characteristics related to solubility of uranium chemical compounds.

The above assessment results in estimated air concentrations of 0.5, 0.046 and 0.015 µg/m^3 respectively for Type S, M and F absorbing uranium compounds from continuous inhalation exposure which would prevent the accumulation in the kidney from exceeding 0.01 µg U /g kidney, which additionally incorporates a combined uncertainty factor of 10 that takes into account inter-individual variability (e.g. sensitive subpopulation, kidney threshold) and uncertainties related to the biokinetic model.

For the purpose of setting an Ontario-wide AAQC, an air limit based on a 50:50 ratio of Type F at 0.015 µg/m^3 and Type M compounds at 0.046 µg/m^3 is considered a reasonable assumption that best represents the various uranium compound mixtures that could be found in Ontario's ambient air. The resulting long-term average air concentration of 0.03 µg/m^3, based on modelled uranium accumulation in the kidney

over a 50-year exposure period, is considered to protect for continuous lifetime inhalation exposure. As such, the MOE has interpreted the results as an annual average concentration. However, under certain conditions, risk of kidney toxicity from uranium exposure at a less-than-lifetime exposures may be of concern. For example, the amount of uranium that is absorbed into the body depends on the route of entry and solubility of uranium compounds, with Type F compounds more readily and quickly absorbed by inhalation than Type M or S. This concern could be addressed site-specifically, where Type F compounds are predominantly emitted.

Finally, the considerations of the biokinetic model are based on estimates of inhaled uranium within the respirable particulate. It is therefore reasonable to base the annual AAQC on the PM10 fraction (particulate matter of less than 10 μm aerodynamic diameter).

Based on an evaluation of the scientific rationale of air criteria from leading agencies, an examination of current toxicological research, and comments from stakeholders, an annual average AAQC of 0.03 μg/m^3,for uranium and uranium compounds in the PM10 size fraction, based on kidney toxicity associated with exposure to these compounds was developed (MOE, 2011d).

Conversion to 24-hour AAQCs: Calculations

Under the MOE's current process, an annual AAQC is converted to a 24-hour AAQC based on the use of conversion factors. These conversion factors were originally derived from empirical data of monitored ambient air levels of sulphur dioxide (SO_2) in urban areas, and also near point sources and atmospheric dispersion modelling of specific sources. The urban ambient air data, acquired in eight of the largest U.S. cities, together with Ontario data available at that time, showed a relationship between a 1 hour average and an annual average exposure at the respective monitoring locations. The MOE used this information to select a conversion factor of 5 to convert from an annual to a 24-hr average, derived from a modified power law relationship (described in more detail in Appendix A).

A detailed discussion of the development of conversion factors, and the empirical data that supports the selection of such conversion factors, is found in Appendix A. Briefly, the conversion factors used by the MOE are derived from a combination of evidence from observed data, meteorological considerations, and guidance from the US EPA.

Peak-to-peak ratios are site-specific, and are typically dependent on the following:

- characteristics of emission sources: source type, source height
- receptor configuration: height of receptor, distance to source
- meteorology: atmospheric stability
- topography

In spite of this site specificity, the MOE established province-wide conversion factors that were reasonably health protective in most ambient air situations. The conversion factors used by the MOE are largely based on a power-law relationship of the form:

$$C_{long} = C_{short} (t_{short}/t_{long})^p$$

Where C_{long}= the concentration for the longer averaging time

                 C_{short}= the concentration for the shorter averaging time

                 T_{short} = the shorter averaging time (in minutes)

                 T_{long} = the longer averaging time (in minutes)

and, p = the power law exponent

For ambient air assessments, the MOE calls for using a value of $p = 0.28$; the rationale for its utilization is discussed in Appendix A.

Thus, in conversions from the annual averaging time to the 24-hour averaging time, the formula would read as follows:

$$C_{annual} = C_{24hr} (t_{24hr}/t_{annual})^{0.28}, \text{ or,}$$

$$C_{annual} = C_{24hr} (24/8760)^{0.28}$$

$$C_{annual} = C_{24hr} (0.19)$$

$$C_{24hr} = C_{annual} / (0.19)$$

In practice, the Ministry uses a factor of 0.2 to convert from an annual averaging time value to a 24-hour averaging time value, so that the latter value is 5x the annual averaging time value. Examples of these conversions are presented below.

Example: Benzene (an annual AAQC of 0.45 µg/m^3(0.14 ppb))

     C_{24hr}= 0.45/0.2

     C_{24hr}= 2.3

Thus, the MOE established a converted 24-hour average AAQC of 2.3 µg/m^3 (0.7 ppb) for benzene, based on carcinogenicity associated with exposure to this compound, and using the effects based annual AAQC of 0.45 µg/m^{3}as a foundation (MOE, 2011b).

Example: 1,3-Butadiene (an annual average AAQC of 2 µg/m^3 (0.9 ppb))

     C_{24hr}= 2.0/0.2

     C_{24hr}= 10

Thus, the MOE established a converted 24-hour average AAQC of 10 µg/m^3 (4.5 ppb) for 1,3-butadiene, based on carcinogenicity associated with exposure to this compound, and using the effects based annual AAQC of 2 µg/m^3 as a foundation (MOE, 2011c).

Example: Uranium (an annual average AAQC of 0.03 µg/m^3)

$$C_{24hr}= 0.03/0.2$$

$$C_{24hr}= 0.15$$

Thus, the MOE established a converted 24-hour average AAQC of 0.15 µg/m^3 for uranium and uranium compounds in the PM10 size fraction, based on kidney toxicity associated with exposure to these compounds, and using the effects-based annual AAQC of 0.03 µg/m^3 as a foundation.

Limitations of the Approach: *Converted 24-hour AAQC*

The converted 24-hour AAQC is not directly linked to an effect and instead provides an indication whether the effects-based annual AAQC may be exceeded. This limitation does not impact this AAQC's use as an air quality target but has been criticized when used to set regulatory air standards for evaluating the contributions of contaminants to air by regulated emitters. MOE's stakeholders have argued that compliance with an air standard should not be evaluated based on a converted value. In response, the MOE introduced annual air standards, for the first time, for six contaminants in 2011. However, since AAQCs are non-regulatory values used to assess general air quality, conversion factors continue to be used to generate converted AAQCs, to evaluate ambient air quality.

As mentioned above, if the converted 24-hour AAQC is met then the annual AAQC is assumed to have been met and no effects are expected over continuous lifetime exposure. The converted 24-hour AAQC is also protective against effects from short-term exposure provided that these other effects do not occur at concentrations less than five times the annual AAQC (i.e., the converted 24-hr AAQC). However, if short-term adverse effects which may occur within 24-hours at levels less than a value equal to 5x the annual AAQC were of concern, then an additional short-term AAQC specific to that other effect would be warranted.

As noted above for approach 1, the converted 24-hour AAQC is not appropriate for assessing the health risk associated with non-continuous exposures, such as those resulting from spills or fires. This gap is filled by the direct development by the Texas Commission on Environmental Quality (TCEQ) of a short-term 24-hr effects-based value. Ontario tends to evaluate such occurrences on a case-by-case basis.

3.0 Comparison of Converted 24-hour AAQCs to TCEQ 24-hr AMCVs

Comparison of the MOE 24-hour AAQCs and the Texas 24-hour AMCVs is made here in the context of the underlying application of these approaches. As noted above, the 24-hour AAQCs set by the MOE are targets for air quality based on protection against effects from continuous lifetime exposures to a contaminant. As long as the 24-hour AAQC is met, then no adverse effects are expected in short-term or long-term exposures. The advantage of the converted 24-hour AAQC is that it provides a single target that is protective of both short-term and long-term exposures (as long as assumptions regarding thresholds for other short-term effects are met). However, since converted AAQCs are not specific to the exposure period or effect being considered, they are not suitable for use in risk assessments, as the converted 24-hour AAQC are not *directly* linked to an adverse health effect.

In comparison, the 24-hour AMCVs set by Texas are intended to evaluate 24-hour monitoring data. Since they are specific to the exposure period and effect being considered, they may be used to assess risk, and theoretically, could be used in risk assessments, as they are directly linked to an adverse health effect. However, they cannot be used in place of the 1-hour or annual values developed for the same contaminant.

As indicated in Table 1, the 24-hour AAQCs for both benzene and 1,3-butadiene are significantly less than their corresponding effects-based TCEQ 24-hour AMCV. This is expected for two reasons: 1) the annual AAQCs set by the MOE uses a lifetime incremental cancer risk target of 10^{-6} in comparison to Texas's target of 10^{-5} (comparing annual AAQC and annual AMCV) and, 2) the converted 24-hour AAQC is intended to be protective against a continuous lifetime exposure scenario, whereas the AMCV is intended as a tool specifically to evaluate an individual 24-hour monitored value. These comparisons illustrate that, for these two contaminants, the conversion factor used to establish the converted 24-hour AAQC is protective against other effects expected in short-term exposures, which was the intent.

Table 1: Comparison of 24 Hour AAQC to the TCEQ 24-hr AMCV (all µg/m^3).

Chemical	Ontario		Texas		
	Annual AAQC [1]	**Converted 24-hr AAQC**	1-hr acute AMCV	**24-hr AMCV**	Annual AMCV [2]
Benzene	0.45	**2.3**	580	**320**	4.5
1,3-Butadiene	2	**10**	3700 (6-hr)	**950**	20

[1] The Ontario annual AAQC is based on a risk target of 10^{-6}

[2] The annual AMCV from Texas is the lower of the chronic non-cancer or cancer value, based on a risk target of 10^{-5}

Appendix A: Development of Conversion Factors

This appendix is based on materials prepared by scientists of the MOE's Environmental Monitoring and Reporting Branch.

Background

The conversion factors used by the MOE are based on the ratios of peak-to-peak air concentrations, and are derived from a combination of empirical evidence from observed data, and guidance from the US EPA.

These peak-to-peak ratios are generally site specific, and are typically dependent on the following:

- characteristics of emission sources: source type, source height
- receptor configuration: height of receptor, distance to source
- meteorology: stability, etc
- topography

In spite of this site specificity, the goal of the MOE was to establish province-wide conversion factors that were reasonably health protective (i.e. conservative) in most ambient air situations.

The conversion factors used by the MOE and various other regulatory agencies across North America are largely based on a power-law relationship of the form:

$$C_{long} = C_{short}\ (t_{short}/t_{long})^p$$

Where　　　C_{long} = the concentration for the longer averaging time

　　　　　　C_{short} = the concentration for the shorter averaging time

　　　　　　T_{short} = the shorter averaging time (in minutes)

　　　　　　T_{long} = the longer averaging time (in minutes)

and,　　　　p = the power law exponent

There are a number of factors influencing selection of values for p, and include atmospheric stability, terrain, surface roughness, source characteristics and location of interest relative to the source.

The exponent on this relationship is typically governed by Pasquill-Gifford (P-G) stability class (Table 1). Briefly, the most commonly used method of categorizing the amount of atmospheric turbulence present was the method developed by Pasquill in 1961. He categorized the atmospheric turbulence into six stability classes named A, B, C, D, E

and F, with class A being the most unstable or most turbulent class, and class F the most stable or least turbulent class. Atmospheric stability refers to processes that contribute to or reduce vertical mixing of pollutants. These processes can include thermal instabilities resulting from surface heating or cold air advection aloft that results in overturning of air, inhibition of thermal instability from, for example, cooling of air near the surface as the earth's heat radiates away overnight, and mechanical mixing as winds blow over obstructions on the surface such as buildings and hills.

Table 2 provides the meteorological conditions that define each class.

Table 1: The Pasquill-Gifford stability classes and averaging time power law exponent (p)

Stability Class	p	Definition
A	0.5	very unstable
B	0.5	unstable
C	0.333	slightly unstable
D	0.2	neutral
E	0.167	slightly stable
F	0.167	stable

Table 2: Meteorological conditions that define the P-G stability classes

Surface windspeed		Daytime incoming solar radiation			Nightime cloud cover	
m/s	Mph	Strong	moderate	slight	>50%	<50%
< 2	< 5	A	A – B	B	E	F
2 – 3	5 – 7	A – B	B	C	E	F
3 – 5	7 – 11	B	B – C	C	D	E
5 – 6	11 – 13	C	C – D	D	D	D
> 6	> 13	C	D	D	D	D

For ambient air assessments, the MOE (2009) calls for using a value of p = 0.28, which is approximately the average of the above values of p under slightly unstable and neutral conditions (P-G stability classes C and D, respectively), and are among the most commonly occurring stability classes in Ontario. However, the regulation does allow for a proponent to request approval to use an alternative value of 'p' if appropriate.

Recall:

$$C_{long} = C_{short} \left(t_{short}/t_{long} \right)^p$$

The use of this relationship and value of "p" results in the following conversion factors (rounded, in some cases):

Averaging Time	Multiplying Factor
1 Hour to 1/2 Hour	1.2
1 Hour to 8 Hours	0.6
1 Hour to 24 Hours	0.4
1 Hour to Annual	0.08
Annual to 24 hour	5
24 to ½ hour	3

The US EPA also uses conversion factors to adjust model predicted maximum air concentrations to other averaging times. The factors are listed in the U.S. EPA document *Screening Procedures for Estimating the Air Quality Impact of Stationary Sources, Revised, EPA-454/R-92-019.* These conversion factors are noted to be "based on general experience with elevated point sources", and limits are provided to the user to be able to adjust the factors (with prior EPA approval) to better suit the actual source characteristics and source-receptor relationships. They are intended to be used as a rough guide for estimating maximum concentrations for averaging times greater than 1 hour, and are as follows:

Averaging Time	Multiplying Factor
3 Hours	0.9 (+/- 0.1)
8 Hours	0.7 (+/- 0.2)
24 Hours	0.4 (+/- 0.2)
Annual	0.08 (+/- 0.02)

The US EPA (1992) states the following about the factor variability:

> The numbers in parentheses are recommended limits to which one may diverge from the multiplying factors representing the general case. For example, if aerodynamic downwash or terrain is a problem at the facility, or if the emission height is very low, it may be necessary to increase the factors (within the limits specified in parentheses). On the other hand, if the stack is relatively tall and there are no terrain or downwash problems, it may be appropriate to decrease the factors.

It should be noted that the U.S. EPA factors are identical to those used by the MOE (based on the Power Law) for 8 hours, 24 hours and Annual averages, and would result in the same "annual to 24-hour" conversion factor (5) as well.

Empirical Data Supports Selection of Conversion Factors

As presented below, an examination of the empirical data (both measurements and modelled results) supports the use of both the relationship and the default factors used by the MOE and the US EPA.

US EPA data

Trout (1978) analyzed several data sets that included ambient monitoring data from power generation facilities in Tennessee and Ohio River Valley in addition to data from urban monitoring networks in eight U.S. cities for 7 different pollutants. The urban network was a representative of a mix of different sources (stack, area & volume) and pollutants. The available data allowed an analysis of peak to peak concentration ratios over different averaging periods (Table 3). As can be seen in the data, there are dramatic differences between the conversion factors for the two power generation networks which the article attributes to potential differences in average stack heights, meteorology, terrain, background concentrations, and location of monitors relative to the sources.

Table 3: Meteorological conditions that define the P-G stability classes

Source Type	Averaging Time	Conversion Factor
Stack (Tennessee)	1 Hour to 24 Hour	0.2
	1 Hour to Annual	0.008
	Annual to 24 Hour	**19.4**
Stack (Ohio River Valley)	1 Hour to 24 Hour	0.2
	1 Hour to Annual	0.04

	Annual to 24 Hour	5.1
	1 Hour to 24 Hour	0.4
Urban (8 cities)	1 Hour to Annual	0.1
	Annual to 24 Hour	4.5

The data show that the monitored data produced conversion factors from the ORV stations that were somewhat consistent with those used by the US EPA & MOE. Overall, differences between the conversion value within and between sites were likely due to a combination of factors:

- Stack height
- Power plant design and operation
- Regional meteorological differences
- Terrain difference and their effect on plume dynamics
- Regional background concentrations
- Locations of monitors

Since the urban conversion factors are based on a mix of different sources and contaminants they are the most representative and comparable to the generic conversion factors used for regulatory purposes in Ontario. These urban conversion factors were, in fact, very consistent with those used by the MOE and US EPA. However, the data indicate that the factor of 5 that use by MOE to convert annual to 24-hour AAQCs is not conservative for comparisons to exposures from mixtures of sources or those other than stack sources.

Ontario ambient data

The MOE examined observed concentration data from the provincial Air Quality Index (AQI) stations in several Ontario cities of different sizes (Windsor, Chatham, Sarnia, London, Hamilton, Toronto, Mississauga, Sault Ste. Marie), from 2006 – 2008, for NOx and SO_2. This was studied in order to develop observed conversion factors similar to those developed for the urban stations by Trout. It should be noted while other criteria contaminants could have been incorporated into the analysis, a number of them (i.e. O_2 and PM2.5) are regional contaminants with ambient concentrations driven by trans-boundary contributions. For this reason, NOx and SO_2 were used for this analysis. The following ratios were observed:

| | | Ratios | |
City	1H - 24H	1H - Ann	24H - Ann
Windsor Downtown	0.3	0.07	4.4
Windsor West	0.3	0.07	5.1
Chatham	0.3	0.07	4.5
Sarnia	0.4	0.06	10.4
London	0.3	0.07	4.7
Hamilton Downtown	0.4	0.07	5.9
Hamilton Mountain	0.4	0.06	6.5
Toronto Downtown	0.3	0.08	4.9
Toronto West	0.4	0.08	4.9
Mississauga	0.4	0.06	4.8
Sault Ste.Marie	0.2	0.04	5.5
AVERAGE	**0.3**	**0.07**	**5.6**

As can be seen above, the conversion factors vary between cities, likely a result of similar inter-location difference as noted above. Overall, they are similar to the conversion factors developed by Trout, and again, are consistent with those used by the MOE and US EPA.

Additional Jurisdiction: Michigan

In contrast to the 5:1 ratio of 24-hour to annual average used by the MOE, the Michigan Department of Environmental Quality (MDEQ) recommends a value of 11:1 (MDEQ, 2010), which would be consistent with a value of $p = 0.4$. Using this value of p results in the ratios of 1-hour average to annual average shown in Table 4.

Table 4. Averaging time conversion factors as outlined by MDEQ..

Avg Time ($_1$)	Avg Time ($_0$)	p	C_1/C_0	Reference
24-Hour	Annual	0.41	11	MDEQ (2010)
1-Hour	Annual	0.43	50	MDEQ(2010)

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