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A Study of the Air Pollution Index Reporting System

FINAL REPORT

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1 Background

The Air Pollution Index (API) Reporting System is an important tool of risk communication. It informs the public of the local level of ambient air pollution, and the potential health risk it would impose, particularly on vulnerable groups such as children, the elderly, and those with existing cardiovascular and respiratory diseases. People use the API to help them make decisions on outdoor activities; for example, schools and sports organizations may check the latest API figures to decide whether outdoor sporting events should be conducted on a certain day. The Hong Kong API has been devised in a similar way to API systems used in other developed countries, although there are variations in the calculation methods.

In June 2007, the Environmental Protection Department (EPD) of the Hong Kong SAR Government commissioned an 18-month study (Agreement No. CE 57/2006 (EP): Review of the Air Quality Objectives and Development of a Long Term Air Quality Strategy for Hong Kong – Feasibility Study), to review Hong Kong's existing Air Quality Objectives (AQO), first established in 1987; and, following the review, to develop a long-term air quality strategy to achieve the updated objectives. This is in response to the Air Quality Guidelines (AQG), published by the World Health Organization (WHO) in October 2005 for worldwide adoption. As the calculation of the Hong Kong API is based on the 1987 AQO currently being reviewed, it is an opportune time to devise an improved API system that serves as an effective tool of risk communication to the general public.

2 Study Objective

To develop an API reporting system for use in Hong Kong, with full justifications and implementation details.

3 Literature Review

We have conducted a literature review of the API reporting systems in various countries, as stipulated in the Tender of this Study.

3.1 Introduction

We reviewed the major air quality index (AQI) or air pollution index (API) systems around the world, including the United States (US), the United Kingdom (UK), Canada, Australia, China, France, Singapore, South Korea, Taiwan, South Africa, Macau, and Hong Kong (Ove Arup, 2007, 1 and 2; websites 1-14). While there are variations among the AQI / API systems developed by different countries or jurisdictions, all of the systems are designed to report the state of the air quality in a specific area or region, and to communicate its associated health risk. AQI / API systems are, in principle, designed to communicate the *short-term* health impact of local air quality to members of the public (Stieb et al, 2005), although in the US system, references are also made to long-term health risks. Health advisories are issued when the air pollution level is high, so that the general population, including susceptible groups, may take the necessary short-term precautions.

3.2 AQI / API Construction

In essence, an AQI or API is constructed to express the levels of one or more air pollutants, over various critical averaging periods, against a reference. The national air quality standards will usually be used as the reference for the index. A network of air quality monitoring stations will be set up to measure ambient concentrations of common pollutants at fixed intervals. Some monitoring stations are located at the roadside to measure street-level concentrations. In places like Hong Kong and Paris, a roadside or traffic index is reported separately from the general AQI / API (websites 3, 6).

3.2.1 Key Air Pollutants

There are variations with respect to the selection of key air pollutants, as individual countries or jurisdictions will seek to include pollutants that pose the most significant impact on their residents (Elshout & Léger, 2006). Air pollutants commonly used in AQI / API include nitrogen dioxide (NO_2), sulphur dioxide (SO_2), ozone (O_3), carbon monoxide (CO), respirable suspended particulate matter (PM_{10}), and lead. Fine suspended particulate matter ($PM_{2.5}$) is chosen in a few places, while in some Australian states, visibility is also incorporated into the AQI / API calculation (website 4). Conversely, pollutants that appear insignificant in a particular country may be omitted from the national AQI / API model. For example, Canada's Air Quality Health Index (AQHI) does not consider the concentrations of SO_2 and CO (Website 2). China's API excludes O_3 from its calculations (website 10).

3.2.2 Averaging Times

Another important aspect in constructing an AQI / API is the choice of the averaging time(s) for each pollutant. As the primary objective of an AQI / API system is to communicate the health risk related to short-term exposure to air pollutants, it would therefore be natural for the system to track pollutant concentrations over a shorter averaging time.

Based on experience around the world, calculation of the AQI / API is usually based on 1-hour, 8-hour, or 24-hour average monitoring data, depending on the pollutants (Ove Arup, 2007, 1 & 2). It is worth noting that while the concentrations of most air pollutants are measured by a shorter averaging time (like the 1-hour average) for AQI / API calculations, particulate matter (PM) is averaged over a 24-hour period. This is due to the lack of scientific evidence with respect to the exposure-response relationship for PM over a one-hour period (Cairncross et al, 2007).

As a result, when PM is the dominant pollutant, the AQI / API system is not responsive enough to reflect a sudden surge in the level of PM, because the index is based on its concentrations averaged over the past 24 hours. There is inevitably a time lag between the rise in concentration recorded at the monitoring stations and the rise in AQI / API readings; this time lag will delay the issuance of health advisories for impending air pollution episodes. An example that highlights this problem is presented in Appendix 1.

One possible approach to tackling this issue is to incorporate estimated pollutant concentrations for future hours into the calculation of the air quality index. The US has been predicting 8-hour ozone levels, based on the correlation between daily maximum 1-hour and 8-hour ozone values, in order to report AQI and health warnings in a more timely manner. Similarly, the AQI for PM is derived from the average of the past 12 hours and the predicted concentrations in the coming 12 hours (USEPA, 2006).

3.2.3 Calculation of AQI / API

Ambient or roadside concentrations for each pollutant, over different averaging times, will be converted into an index value. In general, there are three common methods to achieve this.

The most popular approach is often called the US-based system. Pollutant concentrations for each pollutant are transformed onto a normalised numerical scale of 0 to 500, with an index value of 100 corresponding to the primary National Ambient Air Quality Standard (NAAQS) for each pollutant (USEPA, 2006, website 14).

Places like Singapore, China, Thailand, Malaysia, South Korea, Taiwan, Hong Kong, and Macau designed their AQI / API systems based on the US model. The key reference point of these systems would be the index value of 100, which is based on the short-term air quality standards of the respective jurisdictions. Very often, the index value of 50 is anchored to the long-term air quality standards.

A similar approach is being used in Australia, whereby pollutant concentrations are also being transformed onto a scale. There, however, a linear or proportional scale is used instead of a normalised scale (i.e. a scale which takes the variation into account), and the index is then calculated in direct proportion to the air quality standards or environmental goals (Ove Arup, 2007, 1). Moreover, the scale used in New South Wales is different from the one used in Queensland, Victoria, and Adelaide (in South Australia). In New South Wales, an index value of 50 means that the pollutant concentration is equal to the standard level. For the other states and cities, the index value of 100 carries the same meaning (Ove Arup, 2007, 1; website 4).

The third approach is the banding system, which is more popular in European countries like the UK and France (websites 3, 13). The main deviation is that instead of using an index scale of 0 to 500, a scale of 0 to 10 is being used. For the UK system, this index scale of 10 is further broken down into four bands of 'low' (1-3), 'moderate' (4-6), 'high' (7-9) and 'very high' (10) (website 13). The key reference point for this banding system is the breakpoint value between the 'low' and 'moderate' bands. The lower bound of index value 4 is set to correspond to the UK Air Quality Standards for all pollutants but NO_2. In this case, the 1-hour national standard for NO_2 is 200 $\mu g/m^3$, whereas the lower bound of index value 4 for NO_2 is 287 $\mu g/m^3$ (website 13).

3.2.4 Reporting of AQI / API

Based on one of the above three approaches, concentrations measured over various averaging times at individual monitoring stations will be transformed into air pollution sub-indices (APSI) for each of the pollutants. Normally, the highest of the sub-indices will be taken as the reported AQI / API, and the contributing pollutant will also be specified.

Reporting the air quality as designated by the level of the single worst pollutant has its limitations. In the real world, multiple pollutants affect the health of the community simultaneously, and the conventional approach simply ignores the joint effects of different air pollutants on human health. For instance, we would logically expect a greater impact on health when several pollutants are breaching their respective short-term standards at the same time, as compared to one pollutant reaching an unhealthy concentration level on its own (Cairncross et al, 2007).

However, the simple addition of the health risks of each air pollutant derived from single-pollutant models, as in the case of Canada's AQHI (see section 3.3.1 below), may be an over-representation of the total health effect, by assuming the effects of each pollutant are independent of the others and the total effects are the sum of the individual effects. While some studies have shown that certain pollutants might have synergistic effects, it is not impossible that some pollutants might antagonise the effect of another. How to assess the joint health risks of multiple air pollutants will remain a subject of debate and future research.

In some places, such as China, a different approach is taken whereby the daily average of a pollutant concentration at a monitoring station will be derived from the hourly readings, and a sub-index will then be calculated for that pollutant. The highest sub-index of the most critical pollutant will become the AQI / API of the area (website 10).

For effective communication, descriptors, colour codes, and health advice or warnings are often assigned to specific ranges of AQI / API values. However, there is no universal guideline regarding the wording of the descriptors or health advisories, or on the colour scheme to be used.

3.2.5 Comparing Different AQI / API Readings

Comparing the air quality in different countries using AQI / API readings is always a difficult endeavour. Firstly, arguably few AQI / API systems are identical. Individual country and jurisdictions will design their own systems to report local air quality in the most appropriate way, which means they would choose different air pollutants (those that predominantly affect the local population) and different reporting systems (using an index scale or a banding system).

Secondly, air pollutant concentrations are often measured at different locations within a city that are not directly comparable. For instance, air quality indices representing measurements taken from the ambient air at background stations are very different from those taken from roadside stations, which are influenced by traffic (Elshout & Léger, 2006).

Thirdly, even for the same measured pollutant concentration, different countries may have different interpretations with respect to its health effect and additional health risk (Elshout & Léger, 2006). For example, in France, the worst endpoint ('very poor') of the NO_2 sub-index is 400 $\mu g/m^3$ (website 3). In the UK, the same value is taken as the lower end of the 'moderate' band (website 13).

The AQI / API systems are, in many ways, a gross generalization of a complex mixture of airborne chemicals into a simple index value. The primary purpose for which they are designed is risk communication to the public, rather than comparison between different cities.

3.3 Recent Developments

3.3.1 Air Quality Health Index in Canada

Canada has been using an Air Quality Index (AQI) system to report current and near-term air quality conditions. A scale of 0 to 100 represents air quality conditions ranging from 'very good' to 'very poor' (website 2). An air quality advisory is issued when the calculated sub-indices of the pollutant concentration exceed, for a fixed period of time, an AQI value of 50, at which point the air quality is defined as changing from 'moderate' to 'poor' (website 2).

While the AQI remains a simple tool for communicating the state of the local air, there is little national consistency in how AQIs are reported. The pollution thresholds, the pollutants included in the AQI formulation, and the use of health-based messages vary from one place to another across the country (website 2). Notably, the thresholds used in determining AQI levels and categories are often based on outdated health science, and tend to reflect environmental regulatory imperatives rather than implications for human health (website 2).

In June 2001, the Government of Canada began working with a variety of stakeholders to address the shortcomings of their conventional AQI system, and to devise an effective risk communication tool for *acute* health effects. Inadequacies of the conventional system included (a) its failure to consider the *combined effects of multiple pollutants*; (b) its failure to reflect the *no-threshold concentration-response relationship* between air pollution and health; and (c) its linkage with standards that might be influenced by factors other than health risk (Stieb et al, 2008; Taylor, 2008; website 2).

A new Air Quality Health Index (AQHI) has been designed to help people understand what a certain state of local air quality means to public health. A national pilot programme began in July 2007 for the city of Toronto. At present, the AQHI is available for about ten communities in Canada, including Vancouver and Victoria (website 2).

The AQHI is constructed as the sum of excess mortality risk associated with NO_2, ground-level O_3, and $PM_{2.5}$ at certain concentrations. It is calculated hourly based on 3-hour rolling average pollutant concentrations, and is then adjusted to a scale of 1 to 10. The value of 10 corresponds to the highest observed weighted average in an initial data set, measured in 10 Canadian cities and covering the period between 1998 and 2000 (Stieb et al, 2008; Taylor, 2008).
The scientific foundation for the AQHI is based on the epidemiological research undertaken at Health Canada. Relative risk (RR) values are estimated, based on local time-series analyses of air pollution and mortality (Stieb et al, 2008; Taylor, 2008).

The AQHI index values are grouped into four health risk categories: 'low' (1-3), 'moderate' (4-6), 'high' (7-9) and 'very high' (10+). Health messages customized to each category, for both the general population and the 'at risk' population, will be disseminated (Stieb et al, 2008; Taylor, 2008; website 2).

3.3.2 *Air Pollution Index System in South Africa*

A similar health-based index has been developed in South Africa in a 'dynamic air pollution prediction system (DAPPS) project', which is led by a consortium of four South African partners, including the Cape Peninsula University of Technology (Cairncross et al, 2007). This API system is based on the relative risk of the well-established excess daily mortality associated with short-term exposure to common air pollutants, including PM_{10}, $PM_{2.5}$, SO_2, O_3, NO_2 and CO. A set of relative risks published by the World Health Organization has been used to calculate sub-index values for particulates, SO_2, O_3 and NO_2. For CO, an RR value of 1.04 (for a 10 ppm increment in exposure) was used after Schwartz (1995). O_3 concentrations in the WHO guidelines was used as a reference level for mortality risk, which forms the basis for calculating the concentrations of other pollutants,

A scale of 0 to 10 is used. Incremental risk values for each pollutant are assumed to be constant, and a continuous linear index scale is developed for each pollutant, with RR = 1 at zero exposure. For consistency between pollutant exposure metrics, the exposures that correspond to the same relative risk are assigned the same sub-index value. The final API is the sum of the normalised values of the individual indices for all the pollutants.

The proposed API has been applied to ambient concentration data collected at monitoring stations in the City of Cape Town for testing. However, it is unsure whether the system has been put into any pilot programme in South Africa.

Following the method by Cairncross et al (2007), Sicard et al. (2011), developed an aggregate index using five air pollutants ($PM_{2.5}$, PM_{10}, NO_2, O_3 and SO_2) for the "Provence Alpes Côte d'Azur" (PACA) region, in the South East of France, using $PM_{2.5}$ as a reference instead. This aggregate index will be used in three European sites – Greece (Athens and Thessaloniki), the Netherlands, and PACA region (Sicard et al, 2012).

3.3.3 *Common Air Quality Index in the European Union*

The Common Air Quality Index (CAQI) has recently been developed by the European Union. Three different indices – hourly, daily and annual – present the air quality conditions in European cities in a simple and comparable way. Both background and roadside situations are represented.

The hourly and daily indices are expressed using a 5-level scale, ranging from 0 (very low) to >100 (very high). The calculation is based on concentrations of PM_{10}, NO_2, and O_3, which are the three pollutants that raise major concerns in Europe. The indices reflect EU alert threshold levels or daily limit values as much as possible.

The annual index, on the other hand, provides an overview of the air quality situation in a given city throughout the year, with respect to the EU standards. It is developed to reflect the effect of long-term exposure to air pollution. The annual index is presented as a comparison to the EU annual air quality standards and objectives. If the index value is higher than 1, the limit values of one or more pollutants are not met. If the index value is below 1, on average all the limit values are met.

It is important to note that the CAQI is a standards-based system and is 'designed to give a dynamic picture of the air quality situation in each city but not for compliance checking' (website 7).

4 Comparison of API values using different AQO

A password-protected webpage (http://envf.ust.hk/dataview/apirs) has been created to calculate the would-be index values by adopting the reporting systems from different countries / states / cities, and to compare them with Hong Kong's current API. The systems included here are China's API, Macao's API, Taiwan's Pollution Standard Index (PSI), USA's AQI, South Korea's Comprehensive Air-quality Index (CAI), Ontario's AQI, British Columbia's AQI, New South Wales' AQI, UK's API, Canada's AQHI, and a derivation of Canada's AQHI with FSP approximated as 0.7 × RSP.

In addition, three more sets of indices, each using the Hong Kong API calculation methodology but with different sub-index thresholds – one based on the WHO AQG and the other two, some modifications of the WHO AQG – have also been calculated.

4.1 HK API based on WHO AQG

This is the most direct application of the WHO AQG (numbers shown in red and italics), with linear interpolation below the guideline and linear extrapolation above the guideline (Table 1). For monitoring stations without $PM_{2.5}$ measurements, we estimated the values from PM_{10} by the formula: $PM_{2.5} = 0.7 \times PM_{10}$.

Table 1: Hong Kong APSI as calculated by WHO AQG

APSI*	PM_{10} **24-hr**	SO_2 **24-hr**	NO_2 **1-hr**	NO_2 **24-hr**	O_3 **8-hr**	$PM_{2.5}$ **24-hr**
0	0	0	0	0	0	0
25	10	5	50	20	25	5
50	***20***	10	100	***40***	50	***10***
100	***50***	***20***	***200***	100	***100***	***25***
200	100	40	400	200	200	50
300	150	60	600	300	300	75
400	200	80	800	400	400	100
500	250	100	1000	500	500	125

*APSI: Air pollution sub-index

4.2 HK API based on WHO AQG-NS

As we apply the WHO AQG directly to the APSI thresholds as defined above, SO_2 24-hr becomes the dominant contributing pollutant in general stations most of the time (Table 6). This differs from the findings of most health studies, which suggest that other pollutants (e.g. PM and NO_2) are more important than SO_2 in terms of overall health risk to the public. Hence, another set of APSI thresholds, known in our plots as WHO AQG-NS, was defined by taking away the SO_2 thresholds from the calculation. The thresholds are shown in Table 2. The $PM_{2.5}$ levels for stations without $PM_{2.5}$ measurements were estimated using the same formula as before.

Table 2: Hong Kong APSI as calculated by WHO AQG without SO_2

APSI	PM_{10} 24-hr	NO_2 1-hr	NO_2 24-hr	O_3 8-hr	$PM_{2.5}$ 24-hr
0	0	0	0	0	0
25	10	50	20	25	5
50	***20***	100	***40***	50	***10***
100	***50***	***200***	100	***100***	***25***
200	100	400	200	200	50
300	150	600	300	300	75
400	200	800	400	400	100
500	250	1000	500	500	125

4.3 HK API based on WHO AQG-F

The final index was defined by adding arbitrary hourly thresholds for PM_{10}, $PM_{2.5}$ and O_3 to help minimize the time-delay / phrase shift problem of the currently defined API. The index was identified as WHO AQG-F, and the thresholds are shown in Table 3. In this model, the short-term (1-hour) thresholds were arbitrarily set to be double that of the longer-term (24-hour or 8-hour) thresholds. Again, the $PM_{2.5}$ levels for stations without FSP measurements were estimated by the same formula as before.

Table 3: Hong Kong APSI as calculated by WHO AQG with hourly PM_{10}, $PM_{2.5}$ and O_3

APSI	PM_{10} 24-hr	PM_{10} 1-hr	NO_2 1-hr	NO_2 24-hr	O_3 8-hr	O_3 1-hr	$PM_{2.5}$ 24-hr	$PM_{2.5}$ 1-hr
0	0	0	0	0	0	0	0	0
25	10	20	50	20	25	50	5	10
50	***20***	40	100	***40***	50	100	***10***	20
100	***50***	100	200	100	***100***	200	***25***	50
200	100	200	400	200	200	400	50	100
300	150	300	600	300	300	600	75	150
400	200	400	800	400	400	800	100	200
500	250	500	1000	500	500	1000	125	250

4.4 Comparing the Level of Exceedance

When we calculated Hong Kong's API using methods from different reporting systems and different thresholds, the number of days of exceedance (on which API > 100), as well as the relative significance of each air pollutant as the contributor to the daily maximum, varied from one method to another.

In Table 4, we have listed out the number of days of exceedance based on Hong Kong's current API calculations by contributing pollutants. It is clear that NO_2 has been the major contributor towards non-compliance at roadside stations, while O_3 has been responsible for most of the exceedance days at the general stations. SO_2 never contributed to any instances of exceedance during this period, while $PM_{2.5}$ was not a contributor because we have no AQO for this air pollutant.

Table 4: HK API based on AQO: Number of Days of Exceedance

HK AQO		Number of Days Exceeding HK AQO as the Contributing Pollutant					
		$PM_{2.5}$	NO_2	O_3	PM_{10}	SO_2	Total
2000	General	0	5	6	4	0	15
	Roadside	0	42	0	0	0	42
2001	General	0	7	13	0	0	20
	Roadside	0	35	0	3	0	38
2002	General	0	10	15	0	0	25
	Roadside	0	36	0	0	0	36
2003	General	0	11	18	3	0	32
	Roadside	0	42	0	2	0	44
2004	General	0	8	35	4	0	47
	Roadside	0	71	0	0	0	71
2005	General	0	1	16	5	0	22
	Roadside	0	48	0	1	0	49
2006	General	0	9	12	1	0	22
	Roadside	0	54	0	7	0	61
2007	General	0	4	9	6	0	19
	Roadside	0	79	0	0	0	79

We then compared the level of exceedance based on the API (daily maximum) calculated from four different sets of threshold values, namely the WHO AQG interim target 1 (IT-1) thresholds, the WHO AQG ultimate values, the WHO AQG-NS values, and the WHO AQG-F values.

Table 5 shows that with WHO AQG IT-1, the number of exceedance days increased significantly. Amongst the pollutants, the contributions of NO_2 and $PM_{2.5}$ to non-compliance have also increased significantly, both at the general and roadside stations. On the other hand, the number of exceedance days due to O_3 dropped slightly. SO_2 contributed very infrequently to exceedance, while PM_{10} never contributed at all.

Table 5: HK API based on WHO AQG IT-1: Number of Days of Exceedance

WHO AQG IT-1		Number of Days Exceeding AQG IT-1 as the Contributing Pollutant					
		$PM_{2.5}$	NO_2	O_3	PM_{10}	SO_2	Total
2000	General	27	78	4	0	0	109
	Roadside	75	213	0	0	0	288
2001	General	39	78	4	0	0	121
	Roadside	48	231	0	0	0	279
2002	General	28	58	12	0	1	99
	Roadside	7	202	0	0	0	209
2003	General	49	62	14	0	1	126
	Roadside	20	216	0	0	0	236
2004	General	67	58	26	0	5	156
	Roadside	11	252	0	0	0	263
2005	General	71	51	10	0	3	135
	Roadside	16	225	0	0	0	241
2006	General	54	64	13	0	2	133
	Roadside	20	237	0	0	0	257
2007	General	59	60	7	0	1	127
	Roadside	9	248	0	0	0	257

When we set the WHO AQG's ultimate limits as the threshold values for calculation (Table 6), there were significant changes in the API. Firstly, exceedance occurred almost every day. This was expected, as the threshold values for all pollutants are very stringent under the WHO AQG. Secondly, $PM_{2.5}$ became even more dominant as the contributing pollutant at the roadside stations. Thirdly, SO_2 emerged as a significant contributor to non-compliance with the 24-hour standard, especially at the general stations. Fourthly, the role of NO_2 became overshadowed by the other pollutants. However, it is important to note that the concentrations of NO_2 were still fairly high, even though the gas was relatively less dominating than pollutants such as FSP or SO_2.

Table 6: HK API based on WHO AQG: Number of Days of Exceedance

WHO AQG		Number of Days Exceeding AQG as the Contributing Pollutant					
		FSP	NO_2	O_3	PM_{10}	SO_2	Total
2000	General	144	1	12	23	171	351
	Roadside	308	0	0	6	52	366
2001	General	147	0	6	24	172	349
	Roadside	343	0	0	9	13	365
2002	General	126	0	10	8	206	350
	Roadside	318	0	0	5	42	365
2003	General	164	0	6	10	170	350
	Roadside	305	0	0	22	38	365
2004	General	159	0	1	3	197	360
	Roadside	255	0	0	21	89	365
2005	General	149	0	5	2	197	353
	Roadside	212	0	0	56	97	365
2006	General	147	0	1	4	207	359
	Roadside	207	0	0	53	103	363
2007	General	175	0	2	1	183	361
	Roadside	172	2	0	77	106	357

Tables 7 and 8 show the number of exceedance days by applying the threshold values of WHO AQG-NS (see Table 2) and WHO AQG-F (Table 3). The results were quite similar in both cases, with $PM_{2.5}$ increasing in dominance at the expense of SO_2.

Table 7: HK API based on WHO AQG-NS: Number of Days of Exceedance

WHO AQG-NS		Number of Days Exceeding AQG-NS as the Contributing Pollutant					
		$PM_{2.5}$	NO_2	O_3	PM_{10}	SO_2	Total
2000	General	243	1	16	36	–	296
	Roadside	359	1	0	6	–	366
2001	General	267	0	9	34	–	310
	Roadside	356	0	0	9	–	365
2002	General	259	0	13	22	–	294
	Roadside	360	0	0	5	–	365
2003	General	241	0	12	27	–	280
	Roadside	340	0	0	24	–	364
2004	General	311	0	3	12	–	326
	Roadside	338	0	0	27	–	365
2005	General	295	0	9	6	–	310
	Roadside	286	0	0	79	–	365
2006	General	291	0	8	13	–	312
	Roadside	288	0	0	75	–	363
2007	General	276	0	3	16	–	295
	Roadside	240	2	0	105	–	347

Table 8: HK API based on WHO AQG-F: Number of Days of Exceedance

WHO AQG-F		Number of Days Exceeding AQG-F as the Contributing Pollutant					
		$PM_{2.5}$	NO_2	O_3	PM_{10}	SO_2	Total
2000	General	244	1	14	37	0	296
	Roadside	359	1	0	6	0	366
2001	General	270	0	8	33	0	311
	Roadside	356	0	0	9	0	365
2002	General	260	0	13	21	0	294
	Roadside	360	0	0	5	0	365
2003	General	240	0	11	32	0	283
	Roadside	340	0	0	25	0	365
2004	General	315	0	3	11	0	329
	Roadside	338	0	0	27	0	365
2005	General	306	0	8	6	0	320
	Roadside	288	0	0	77	0	365
2006	General	292	0	6	17	0	315
	Roadside	286	0	0	77	0	363
2007	General	280	0	3	19	0	302
	Roadside	243	2	0	103	0	348

5 Modelling Hospital Admissions Data using the Canadian Approach

5.1 Rationale for the Use of the Canadian Model

In the literature review, we found that the AQHI adopted by Canada (Stieb et al, 2008) was a unique, health-based system that made use of local health data, established a link with air pollution data, and then calculated the impact of different levels of air pollution on a specific health outcome (mortality). The purpose was to ensure that the API reporting system will be based on health outcomes observed locally, instead of on study findings in other countries, where the quantitative relation between air pollution and health might be different and therefore not directly applicable. Another feature of the Canadian AQHI system was that it combined the effects of multiple pollutants, assuming them to be independent and hence additive. The use of a daily maximum of the 3-hour moving average in the construction of the statistical model was a compromise between timeliness (using real-time data) and the delayed, cumulative effects of continuous exposure to air pollution.

5.2 Statistical Modelling

We modified the Canadian system by substituting mortality data – which are less sensitive indicators of health – with emergency hospital admissions for respiratory and cardiovascular diseases. The advantage of using this indicator of ill-health is that we have a comprehensive and uniform dataset in the public hospitals. The data are subject to stringent quality control, and represent over 90% of all emergency hospital admissions throughout Hong Kong. We used local data to obtain relative risks (RR) for individual air pollutants. Assuming a linear dose-response relationship, and a zero excess risk when the air pollutant concentration reaches zero (rather than using an arbitrary standard like the WHO AQG as the zero excess risk reference), we calculated the proportion of excess emergency hospital admissions for respiratory and cardiovascular diseases that were attributable to air pollution at different levels of air pollution.

To estimate the RR, which quantifies the risk of hospital admissions for different air pollutants, either singly or in their joint effects, we performed a time series study using Poisson regression. Data on hospital admissions for respiratory and cardiovascular diseases (from 2001 to 2005) were obtained from the Hospital Authority. Daily meteorological variables (mean temperature and humidity) were obtained from the Hong Kong Observatory. The statistical model chosen was a generalized additive model, one that has been most widely used in the current literature. Daily emergency hospital admissions for respiratory and cardiovascular diseases were used as the health outcome variables in the model, and smoothing for the time variable was done for various degrees of freedom using smoothing splines.[*] The model was adjusted for daily mean temperature and relative humidity, a 'day of the week' indicator, a holiday indicator, and a season indicator as potential confounders. Over-dispersion was adjusted by the quasi likelihood method and auto-correlation was adjusted by adding auto-regressive terms into the core model. Residuals plots and PACF plots were used to examine the goodness of fit of the model. Hourly air pollutant concentrations were provided by the Environmental Protection Department, and the maximum of the 3-hourly moving average of a day was used to define the daily concentrations for each air pollutant. The model was tested for the lag effect of air pollution, on the same day (lag day 0), lag day 1 (air pollutant concentration on the previous day) and lag day 2 (two days ago). The 'best lag day' for each air pollutant was chosen according to the maximum t-value, calculated using the gam.exact function of S-PLUS (iHAPSS, 2002; Dominici et al, 2004).

There is no consensus in the literature on the choice of models for hospital admissions. For mortality data, investigators of the National Morbidity, Mortality, and Air Pollution Study (NMMAPS) used 7 degrees of freedom per year (Peng et al, 2005).

The percentage of excess daily hospital admissions, also known as the 'percentage excess risk' (%ER), was expressed as: $\Sigma_{i=1,\ldots,p}\ (e^{\beta i x ij}-1)\times 100\%$. β_i was the regression coefficient of pollutant *i* from the time series analysis, and x_{ij} was the concentration of pollutant *i* at time *j*, for a total of p pollutants. The %ER was calculated using the regression coefficients of four pollutants, NO_2, O_3, PM_{10}[§] and SO_2[**], and their respective concentrations on each day over the five-year period. An equal weight was assumed for each pollutant.

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[*] In the generalized additive model, the degrees of freedom (df) for the smoothing parameter tested were: 0, 10, 20,….to 160. RR estimates varied with varying df, and the RR values peaked at different df. NO_2 peaked at df=50; $PM_{2.5}$, at df=60, and O_3 at df=80, while SO_2 peaked at df=0. The model with 70 degrees of freedom was chosen in favour of the model with the minimum AIC (df=147) because the RRs of most air pollutants (except SO_2) were near their maximum values. For SO_2 , the RR was much smaller than all the other pollutants and contributed little in the calculation of the %ER. At df=70, the relative weights of the RRs of all pollutants in their contribution to the %ER are much more balanced than that using RRs derived in the "statistically best-fit model", where the relative weightage of O_3 is much higher than all other pollutants. The relative weightage of the beta values for NO_2, O_3, SO_2 and PM_{10} at df=70 were: 32.4%:37.1%:10.1%:20.5%. At df=147, the relative weightage for NO_2, O_3, SO_2 and PM_{10} were: 25.9%:51.4%:6.0%:16.7%. This implies the %ER will be dominated by O_3 concentrations while other pollutants' contributions to the %ER are insignificant. The rationale for our model choice is to estimate the highest levels of risk that can be attributed to most air pollutants, while maintaining a more balanced contribution of %ERs among the 4 pollutants.

[§] It is generally recognized that $PM_{2.5}$ penetrate the lung more deeply than PM_{10} do. However, in the calculation of excess hospital admissions, we have chosen PM_{10} instead of $PM_{2.5}$, because data on PM_{10} were available in all air monitoring stations, compared to $PM_{2.5}$, with data limited to a few stations only. The RR for PM_{10} is therefore more robust than that for $PM_{2.5}$. It is anticipated that more comprehensive monitoring of $PM_{2.5}$ will be implemented by EPD in the future. In Hong Kong, $PM_{2.5}$ is strongly correlated with PM_{10}, with a high $PM_{2.5}$ to PM_{10} ratio of about 0.7.

[**] The Canadian AQHI excluded SO_2 from its model.

Similar models were constructed for hospital admissions for children below 5 years of age and for those aged 65 years and above, and the corresponding %ERs were estimated using the respective β for the pollutants obtained from these models, as described above.

To test the validity of the model, we performed sensitivity analyses by adding an indicator for influenza (Wong et al, 2002), and by splitting the 5 years time series into two (2001 – 3 and 2004 – 5) to estimate the RRs of the respective air pollutants.

5.3 *Banding of the Excess Risk of Hospital Admissions Attributable to Air Pollution*

The excess risks were categorized into five bands, in terms of the risk level from short-term exposure to air pollution: Band 1 (low risk), Band 2 (moderate risk), Band 3 (high risk), Band 4 (very high risk) and Band 5 (serious risk). This banding used the short-term exposure limit values of the four air pollutants, as recommended by the World Health Organization's Air Quality Guidelines 2005§ (with some modifications for NO_2)[††], as reference points to identify a 'very high risk' band – one where the general public is exposed to a significant health risk. The rationale is that on a day with concentrations of air pollutants at the respective levels, the sum of %ER will be considered as the threshold above which the risk is too high.

To address the health risk to the vulnerable groups – children aged under 5, and the elderly aged 65 years and above, the reference point for the %ER of these groups was further adjusted. An adjustment factor was derived from the ratio of the median %ER for children under 5 or those aged 65 years and above (whichever was the larger) to that for all ages. The adjusted %ER, used as a limit for short-term exposure to air pollutants by the high risk groups, was obtained by dividing the %ER for all ages by the adjustment factor. Half of this adjusted %ER value was arbitrarily used as a dividing line between the 'low risk' category ($\leq$0.5 $\times$ adjusted %ER) and the 'moderate risk' category (>0.5 $\times$ adjusted %ER to $\leq$ adjusted %ER). The %ER above the adjusted %ER up to the unadjusted %ER was categorized as 'high risk', as the air pollutant concentration would pose a significant health risk to the high-risk age groups but not to the other age groups. When the %ER is above the unadjusted %ER value, the health risk was categorized as 'very high', because the air pollutants would pose a significant health risk to people of all ages. A %ER 50% higher than the unadjusted %ER value is labelled as 'serious'.

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§ The WHO Guidelines set the 24-hr mean for $PM_{2.5}$ and PM_{10} as the values representing the 99th percentile of the distribution of daily values, based on the relation between the daily mean and the respective annual mean AQGs. The 8-hr mean for O_3 and one-hour mean for NO_2 were derived from studies of short-term health effects, including time series and toxicological studies. The 24-hr AQG for SO_2 was based on an intervention study and a Hong Kong-London comparative study, but not on the relation between the distribution of the daily and annual mean concentrations, with an unusual, identical AQG for 24-hr and annual SO_2.

†† The WHO AQGs for short-term exposure are: 200μg for NO_2 (one-hour), 100μg for O_3 (8-hour mean), 50μg for PM_{10} (24-hour) and 20μg for SO_2. Since the averaging time for NO_2 was one hour, we calculated the corresponding value of the NO_2 concentration for a 3-hour moving average, by regressing the one hourly concentrations with the 3-hour moving average in a linear regression model using NO_2 data in our study period. The corresponding value was 184.45μg/m^{3}with a 95% lower confidence limit of 129.8μg/m^3. The lower 95% confidence limit of 129.8μg/m^3 was used as the concentration of our calculation of %ER for NO_2.

6 Results

6.1 Air Pollutants and Emergency Hospital Admissions for Cardio-Respiratory Diseases

All five air pollutants were significantly associated with emergency hospital admissions for respiratory and cardiovascular diseases for all age groups combined: NO_2, O_3, PM_{10}, $PM_{2.5}$ and SO_2. Their respective RRs were: 1.0045, 1.0051, 1.0028, 1.0022 and 1.0014 (Table 9). The RRs were significant at $p<0.0001$ for the first four pollutants, and $p=0.0131$ for SO_2. The 'best' lag day was lag day 0 (same day) for all pollutants except O_3 (where lag day 1 was the 'best lag day').

The core model (before the air pollutant concentration was added) is shown as follows:

Log (resp card 0) = resp card 1-7 + s (day, 70) + s (humidity, d.f. = 15) + s (temperature, d.f. = 15) + day of week indicator + season indicator + holiday indicator

The residual plots did not show any obvious cyclical patterns (See Appendices 4 and 5). The PACF plots showed that autocorrelation was insignificant up to lag day 12 (Figure 2 of Appendix 6).

Explanatory note:

The dependent (outcome) variable, resp card 0, is the daily number of emergency hospital admissions for respiratory diseases and cardiovascular diseases.

The independent variables are:

Resp card 1 + resp card 2 + resp card 3 + resp card 4 + resp card 5 + resp card 6 + resp card 7 are the numbers of emergency hospital admissions for respiratory diseases and cardiovascular diseases from lag day 1, day 2, … to day 7. They are also called auto-regressive terms.

s (day, 70) is the time (or day) variable and is smoothed with 70 degrees of freedom (d.f.).

s (humidity, d.f. = 15) is the daily mean humidity and is smoothed with 15 d.f.

s (temperature, d.f. = 15) is the daily mean temperature (in Celsius) and is smoothed with 15 d.f.

Day of week indicator shows the day of the week variable (Monday, Tuesday, …, Sunday).

(Cold) season indicator takes the value 1 from December to February and 0 during the period from March to November.

Holiday indicator takes the value of 1 on public holidays.

The RRs for high risk groups, namely those aged 65 years and above, and children under 5 years, were estimated using the same model. Compared to RRs for all ages, those aged 65 years and above had higher RRs for NO_2, PM_{10}, O_3 and SO_2, but slightly lower RR for $PM_{2.5}$. All RRs were significantly higher than one. The RRs for children under 5 years were even higher for O_3, $PM_{2.5}$ and SO_2, but lower than the RR for all ages for PM_{10} and NO_2. The RRs were significant for O_3 and NO_2, but insignificant for $PM_{2.5}$ and SO_2.

Table 9: Relative risk of hospital admissions for cardiovascular and respiratory diseases per 10 μg/m^3 increase in air pollutant concentrations

	RR (95% CI) per 10 μg/m^3 increase in air pollutant concentration (single pollutant model)				
Emergency hospital admissions	NO_2	O_3	PM_{10}	$PM_{2.5}$	SO_2
Cardiovascular and respiratory (all ages) (d.f.=70)	1.0045§ (1.0044-1.0046) (lag day 0)	1.0051§ (1.0050-1.0052) (lag day 1)	1.0028§ (1.0027-1.0029) (lag day 0)	1.0022§ (1.0021-1.0023) (lag day 0)	1.0014* (1.0013-1.0015) (lag day 0)
#: Cardiovascular and respiratory (≥65 years) (d.f.=70)	1.0051§ (1.0039-1.0063) (lag day 0)	1.0057§ (1.0045-1.0069) (lag day 1)	1.0033§ (1.0028-1.0044) (lag day 0)	1.0020** (1.0009-1.0032) (lag day 2)	1.0017* (1.0003-1.0030) (lag day 0)
Cardiovascular and respiratory: (<5 years) (d.f.=70)	1.0034** (1.0032-1.0037) (lag day 2)	1.0074§ (1.0072-1.0077) (lag day 0)	1.0025* (1.0003-1.0048) (lag day 2)	1.0025 (NS) (0.9999-1.0051) (lag day 1)	1.0019 (NS) (0.9991-1.0046) (lag day 1)

The ≥65 years age group constituted about 80% of all respiratory and cardiovascular admissions.

* p<0.05; ** p<0.001; § p<0.0001; NS = not significant at p=0.05; d.f. = degree of freedom for the variable 'days'

6.2 *Sensitivity analysis*

To examine the effect of influenza on hospital admissions, we added an indicator variable on to the model using an arbitrary definition of an influenza week, as one during which the number of influenza hospital admissions exceeded the 75th percentile for the year (Wong et al, 2002). The differences in RR from that in our original model ranged from -0.0013% to -0.0131%, which had little effect on our calculation of % excess risks. There was little change in the statistical significance of the RRs (see Appendix 7). To test the stability of the model, we split the time series into 2 periods: 2001 – 03 and 2004 – 04 and ran separate models. All the RRs were similar to that in the original model, with differences ranging from -0.0061% to 0.144%, for the 2-year model, and from -0.0195% to 0.0796% for the 3-year model. The 95% confidence intervals of the RRs in the split models were wider, but remained statistically significant, except for SO_2 in the 3-year model (see Appendix 7).

6.3 *Excess Risks of Hospital Admissions Attributable to Air Pollution*

The frequency distribution of the daily excess risk of hospital admissions attributable to air pollution (expressed as a percentage, %ER) during the time period is shown in Figure 1. During the five years study period, the minimum of the percentage of excess daily hospital admissions attributable to air pollution (% ER) was 2.64%; the maximum was 31.51%, with a median of 9.04% and a mean of 9.50%.

6.4 *Excess Risks of Hospital Admissions Attributable to Air Pollution Among High-Risk Groups*

The % ER for hospital admissions for cardiovascular and respiratory diseases were calculated for those aged 65 and above (with a minimum % ER of 3.02%, a median of 10.34%, a maximum of 36.25% and a mean of 10.86%) and children under 5 years of age (with a minimum % ER of 2.59%, a median of 9.44%, a maximum of 33.32% and a mean of 10.01%).

6.5 *Health risk categories and AQHI Bands*

The 'anchor point' of the %ER that separated 'very high' from the 'high' band was derived from the sum of %ER values calculated from the recommended short-term exposure limit values of the WHO AQG for four pollutants: NO_2 (modified), O_3, PM_{10}[*] and SO_2. These concentrations were: 129.8μg/m^3 for NO_2 (see footnote of section 5.3), 100 μg/m^3 for O_3 (8-hour mean), 50μg/m^3 for PM_{10} (24-hour mean), and 20μg/m^3 for SO_2. Using these values, a %ER of 12.91% was obtained for all age groups. Above this %ER, the AQHI was considered unsafe even for healthy persons in the community and was labelled 'very high'. A %ER above 19.37% (50% higher than 12.91%) was labelled as 'serious'.

For the high risk age groups – comprising both children aged under 5, and the elderly aged 65 years and above – the %ER was adjusted downwards by a factor of 1.144.[‡‡] The %ER of 12.91% was divided by 1.144, giving a value of 11.29%. The %ER of 11.29% was used as an upper limit for short-term exposure to air pollutants by the high risk age groups. Above 11.29%, up to 12.91%, the air pollutant concentration was considered to pose a significant health risk to the high risk age groups, but not to the general population. This range of %ER was categorized as 'high'.

Half of 11.29%, i.e. 5.64%, was used as the cut-off point of 'low risk'/ 'moderate risk' category. The %ER in this range (>5.64%, up to 11.29%) was labelled as 'moderate, while the %ER at 5.64% or below was labeled as 'low'.

The %ER in the categories low, moderate and very high were further sub-divided into equal thirds, making a total of 10 bands from 'low' – 1 to 'very high' – 10. The category 'serious' is labelled as band 10+.

The distribution of the %ER in the 5-year study period is shown in Table 10 and in Figure 1. The health advice corresponding to each band is shown in Table 11.[§]

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[*] PM_{10} was used in the calculation of %ER instead of $PM_{2.5}$ because data on the former are more comprehensive. In addition, the concentrations of PM_{10} increase to a much greater extent than that of $PM_{2.5}$ during dust storm episodes, making PM_{10} a better indicator of health impact on these days.

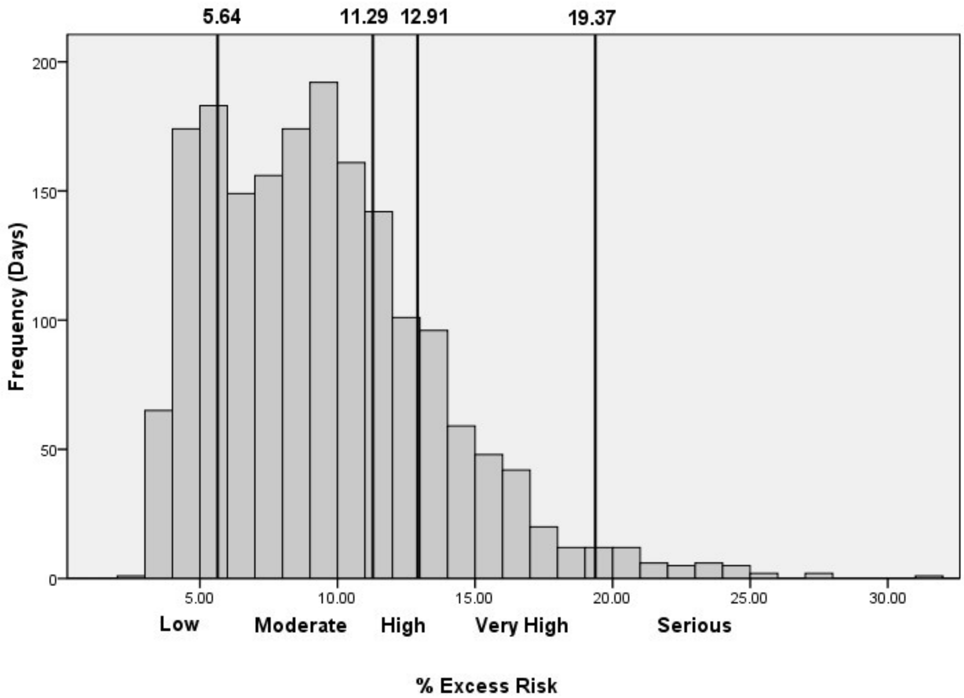
[‡‡] The median %ER for the elderly was 10.34%, which was higher than that for the children (at 9.44%). The %ER for all ages was 9.04%. The ratio of the median %ER for the elderly to that for all ages was 10.34/9.04, or 1.144. The ratio of the median %ER for the children to that for all ages was 1.044. Hence, the adjusted %ER was 11.29% was used, as it was lower than that for children (at 12.37%).

[§] In response to demand from different sectors of the community, the health advice included persons with cardiovascular and / or respiratory diseases and outdoor workers, in addition to children and the elderly, and the general public. It should be noted that the advice was not based on the risk estimates from the results of this study. Advice for persons with cardiovascular / respiratory diseases generally follows that for children and the elderly, whereas outdoor workers were assumed to be healthy, non-elderly adults.

Table 10: Distribution of % excess risk (% ER) of hospital admissions for cardiovascular and respiratory diseases by health risk category and AQHI band

Recommended health risk category	AQHI band	%ER	No. of days	Frequency (%)
Low	1	0 - 1.88	0	0.0
	2	>1.88 - 3.76	36	2.0
	3	>3.76 - 5.64	333	18.2
Moderate	4	>5.64 - 7.52	277	15.2
	5	>7.52 - 9.41	339	18.6
	6	>9.41 - 11.29	306	16.8
High	7	>11.29 - 12.91	194	10.6
Very high	8	>12.91 - 15.07	172	9.4
	9	>15.07 - 17.22	93	5.1
	10	>17.22 - 19.37	27	1.5
Serious	10+	>19.37	49	2.7
Total			1826	100.00

Figure 1: Distribution of % excess risk (%ER) and the categories of health risk



<u>Table 11: Air Quality Health Index and Associated Health Advice</u>

Health Risk	AQHI band	%ER	(i) People who are sensitive to Air Pollution		(ii) Outdoor Workers*	(iii) General Public
			(a) People with existing heart or respiratory illnesses	**(b) Children and the Elderly**		
Low	1	0 - 1.88	No response action is required.	No response action is required.	No response action is required.	No response action is required.
	2	>1.88 - 3.76				
	3	>3.76 - 5.64				
Moderate	4	>5.64 - 7.52	No response action is normally required. Individuals who are experiencing symptoms are advised to **consider reducing** outdoor physical exertion.	No response action is required.	No response action is required.	No response action is required.
	5	>7.52 - 9.41				
	6	>9.41 - 11.29				

* The advice applies to outdoor workers who do not belong to (i).

Health Risk	AQHI band	%ER	(i) People who are sensitive to Air Pollution		(ii) Outdoor Workers*	(iii) General Public
			(a) People with existing heart or respiratory illnesses	**(b) Children and the Elderly**		
High	7	>11.29 - 12.91	People with existing heart or respiratory illnesses (such as coronary heart disease and other cardiovascular diseases, asthma and chronic obstructive airways diseases including chronic bronchitis and emphysema) are advised to **reduce** outdoor physical exertion, and to **reduce** the time of their stay outdoors, especially in areas with heavy traffic. They should also seek advice from a medical doctor before participating in sport activities and take more breaks during physical activities.	Children and the elderly are advised to **reduce** outdoor physical exertion, and to **reduce** the time of their stay outdoors, especially in areas with heavy traffic.	No response action is required.	No response action is required.
Very High	8	>12.91 - 15.07	People with existing heart or respiratory illnesses are advised to **restrict** outdoor physical exertion, and to **restrict** the time of their stay outdoors, especially in areas with heavy traffic.	Children and the elderly are advised to **restrict** outdoor physical exertion, and to **restrict** the time of their stay outdoors, especially in areas with heavy traffic.	Outdoor workers are advised to **reduce** outdoor physical exertion, and to **reduce** the time of their stay outdoors, especially in areas with heavy traffic. Employers are advised to assess the risk of outdoor work, and take appropriate preventive measures to protect the health of their employees.	The general public is advised to **reduce** outdoor physical exertion, and to **reduce** the time of their stay outdoors, especially in areas with heavy traffic.
	9	>15.07 - 17.22				
	10	>17.22 - 19.37				
Serious	10+	>19.37	People with existing heart or respiratory illnesses are advised to **avoid** outdoor physical exertion, and to **avoid** staying outdoors, especially in areas with heavy traffic.	Children and the elderly are advised to **avoid** outdoor physical exertion, and to **avoid** staying outdoors, especially in areas with heavy traffic.	Outdoor workers are advised to **restrict** outdoor physical exertion, and to **restrict** the time of their stay outdoors, especially in areas with heavy traffic.	The general public is advised to **restrict** outdoor physical exertion, and to **restrict** the time of their stay outdoors, especially in areas with heavy traffic.

Health Risk	AQHI band	%ER	(i) People who are sensitive to Air Pollution		(ii) Outdoor Workers*	(iii) General Public
			(a) People with existing heart or respiratory illnesses	(b) Children and the Elderly		
					Employers are advised to assess the risk of outdoor work, and take appropriate preventive measures to protect the health of their employees	

Note:

1. As the health effects on individuals may vary, you should seek advice from a medical doctor if you are in doubt or feel uncomfortable. If you are suffering with existing heart or respiratory illnesses (such as coronary heart disease and other cardiovascular diseases, asthma and chronic obstructive airways diseases including chronic bronchitis and emphysema), you should follow your doctor's advice on the amount of physical exercise and the management of your illness under different air quality health index bands. If you are a smoker, you should quit smoking now!

2. Outdoor workers need to be aware of the potential impact on their health at times when the AQHI reaches “Very High” or “Serious” health risk, and seek advice from a medical doctor if they are in doubt of their health condition or suffer from any chest or breathing discomfort. They should inform their employers of the medical advice so that suitable work arrangements can be worked out.

3. The amount of physical exercise that should be performed differs according to the individual’s physical capacity, and should be tailored to one’s own physical condition. Ask your doctor for advice.

6.6 *Interaction of Air Pollutants with the Cold Season*

We have further examined the interaction between the cold season and the effects of air pollutants, i.e., whether the latter have different effect size (RR) during warm and cold seasons. When interaction terms (season × air pollutant concentration) are added into the model, only O_3 has significant interaction with season, its effect being accentuated in cold seasons (arbitrarily defined from December to March). As a result, the contribution by O_3 to % AR is greater in winter months than in non-winter months. We also tested for interaction between different air pollutants by adding interaction terms. All were statistically insignificant and were excluded in the final model.

6.7 *Annual Air Quality Index (AQI)*

To communicate the health risks caused by long-term exposure to air pollutants, we recommend to publicize Annual Air Quality Indices for those air pollutants, i.e., NO_2 and PM_{10} or $PM_{2.5}$, with annual WHO AQGs, which is similar to EU's Common AQI System. The index is derived from the ratio of the annual mean concentration of an air pollutant to that of the corresponding WHO annual AQG. An annual AQI of one means that the air pollutant concentration is equal to the WHO annual AQG level; an index greater than one would indicate that the health risk resulting from long-term exposure to an individual air pollutant is higher than that caused by exposure to the WHO reference value, whereas an index below one means the opposite. The annual AQI should be calculated on an annual basis and accessible via a hyperlink in the EPD website for AQHI.

7 Discussion

Based on the Canadian methodology, we have calculated the excess risks of hospital admissions for cardiovascular and respiratory diseases that are attributable to air pollution. There are several important assumptions in this model. First, we have assumed that the risk is linear and without a threshold. There is much epidemiological evidence in support of this assumption, especially for the effects of particulates and ozone (WHO, 2005). Another assumption is that each of the four air pollutants contributes independently to a risk to health, with the respective contribution being represented by the RR obtained in a single pollutant model. The additive property of the risks cannot be verified, because of collinearity of the air pollutants (i.e. the concentrations of some pollutants are highly correlated, and not independent of each other). Collinearity is an intrinsic characteristic because some of the pollutants share common sources. Consequently, their individual RR cannot be ascertained reliably in a multi-pollutant model. Nevertheless, an assumption that each air pollutant independently exerts some effect on health is a reasonable one. The additive model, where each air pollutant contributes partially to the overall health risk, is an improvement from the current API where only one air pollutant – whichever happens to exceed the AQO the most – is considered at any given time. We also explored the potential synergistic effects of air pollutants. All interaction terms in the model were statistically insignificant and had little effects on our risk estimates. The concept of additive health risk was also used in the development of air pollution indices by Cairncross et al (2007) and Sicard et al (2011). In both studies, the cut-off point between two indices or bands was arbitrarily chosen as the concentration of one air pollutant as reference, and the RR so derived was then applied to other air pollutants to obtain the equivalent concentrations that were assigned to the same 'index' or band. In the study by Cairncross (2007), the UK standard of 100 μg/m^3 for 1-hour O_3 was chosen for the cut-off between '2 and '3' ('3' is the highest index of the 'low' band.). In Sicard's study (2011), the WHO AQG for 24H PM_{10} of 50 μg/m^3 was used. as a cut-off point between index '3' (low) and '4' (moderate). The major advantage of the AQHI is that it is sensitive to changes in the concentration of any of the four air pollutants. It is noteworthy that summing up the excess risk attributable to all four air pollutants, each derived from the RR obtained from a single pollutant model (which does not take into account, and therefore includes risks from the other pollutants) might have over-estimated the total excess risk.

One major strength of the AQHI is that our health risk is derived from RRs using local health statistics and air pollution data, instead of RRs published elsewhere. Time series models have been extensively used for estimating RRs of short-term exposure to air pollutants on health. An important feature of this approach is that risk factors that do not change on a daily basis, for example smoking prevalence, need not be adjusted, even though smoking is a strong personal risk factor for cardiorespiratory diseases (Schwartz, et al, 1996). Our model has been tested by splitting the series into two periods, and by adding an indicator for influenza. Sensitivity analyses have shown that our model is robust – the RRs are similar with or without adjustment for influenza, and whether or not the models are split.

The β-values we obtained were about 50% to 70% that of the β-values obtained using 24-hour mean. The ratios of the β from 3-hour moving means to the 24-hour means were 0.57, 0.70, 0.50 and 0.57 for NO_2, PM_{10}, O_3 and SO_2 respectively. The β ratios (3-hour to 24-hour) in Stieb's paper were: 0.56, 0.36 and 0.72 respectively for NO_2, PM_{10} and O_3 respectively (β for SO_2 was not significant in that study). Our findings are similar to that reported by Stieb et al (2008) in this respect. Nevertheless, the RRs in our study cannot be directly compared with that in Stieb's paper (which measured mortality risk) or indeed with RRs published elsewhere, owing to the differences in the averaging time of the air pollutants used in model building. Most studies use the 24-hour mean concentration of air pollutants.

One major departure from the Canadian approach is that instead of setting mortality as the health outcome, we have used emergency hospital admissions for cardiovascular and respiratory diseases, which we believe is an improvement.[*] There are several reasons for our choice. Firstly, mortality is dominated by the elderly population, and represents only the 'top of the pyramid' in the spectrum of health outcomes associated with air pollution. It does not reflect the risk to other vulnerable groups, such as young children. By contrast, hospital admissions represent a broader section of the 'pyramid', as they cover a wider age range and affect a greater number of people. While children are at low risk of death, they have been shown to be highly susceptible to air pollution-related illnesses, with even higher % AR for hospital admissions than those aged 65 and above for O_3 and SO_2. Most importantly, the hospital data that are provided by the Hospital Authority (HA) for our study are comprehensive, standardized (with a uniform disease coding system), and of high quality. Over 90% of emergency hospital admissions in Hong Kong are captured in HA hospitals. Hence, our data represent a large majority of the Hong Kong population.

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[*] For comparison, we calculated RRs for cardio-respiratory mortalities for all ages, using Hong Kong-wide data from the Census and Statistics Department. Only NO_2 was significantly associated with mortality. By contrast, all four air pollutants were found to be significantly associated with hospital admissions, and contributed to the %ER. Hence we consider that risk estimates based on hospital admission data better represent the health impact of the general population.

Although our final model uses PM_{10} as one of the four major pollutants, we have compared the results using $PM_{2.5}$ instead. While the results are similar, the reason for advocating the use of PM_{10} is that it can be readily calculated from data available in all air monitoring stations. The AQHI derived from PM_{10} can also provide a good estimate of health risk in the event of dust storms, where the predominant pollutant is PM_{10} rather than $PM_{2.5}$.

Although the addition of an interaction term (O_3 concentration × seasonal indicator) in the model should have better reflected the health risk, the inclusion of a seasonal indicator in the calculation of the % AR would have resulted in an artificial change in its values during the two transitional days between 'hot' to 'cold' seasons and vice versa (specifically, 30 November to 1 December, and 30 March to 1 April). We consider that the disadvantage of using this method to calculate % AR outweighs its advantage. As a confounding variable, both daily mean temperature and season have been adjusted in the model during the derivation of all the RRs. Hence, the temperature and seasonal effects have already been taken into account. The use of a single RR for O_3, regardless of season, is a simplified approach for the calculation of the % AR.

The derivation of the roadside AQHI is limited by the relative paucity of data on O_3 throughout the study period. Roadside levels of O_3 have not been monitored until recently, because of the interaction between O_3 and NO_x. We suggest that roadside O_3 concentrations should be made routinely available in roadside stations for the calculation of the respective % AR.

The ratios of the RR of one air pollutant to the RRs of other air pollutants become substantially different when a different degree of freedom is used in the generalized additive model. We have examined the changes in the RR estimates using different degrees of freedom, and obtained different results on the relative contribution of % AR from different air pollutants, when the degree of freedom for the 'day' variable is altered. There is no consensus on the choice of model. The Air Pollution and Health: a European Approach (APHEA-2) study uses a model with the minimal AIC, i.e., the statistically 'best fitting' model (Atkinson et al, 2001). Some advocate the use of 'a priori' criteria for the degree of freedom, while others have suggested the use of alternative parameters (Peng et al, 2005i). For the AQHI estimate, we have chosen the most conservative model, where the RRs of most pollutants are at or near their largest values. This choice ensures that the risk of air pollution to health will not be underestimated.

By using the concentrations of the four air pollutants in the estimate of the % AR, we are able to estimate the magnitude of the additional risk of hospital admissions on an hourly basis. This is a considerable advantage to our approach in terms of communicating the health risk of short-term exposure to air pollution. We have used the 3-hour moving average of the concentrations of air pollutants in calculating the AQHI, in accordance with the Canadian approach. By contrast, an API system that depends on a 24-hour mean concentration of a pollutant such as PM_{10} would inevitably suffer from a much longer time lag. Another strength to our method is that we have used local health and air pollutant data in deriving the RR. As RRs for air pollutants are known to vary between communities, our approach provides more realistic estimates of health risk from short-term exposure to air pollution.

The banding system of the Canadian approach is not related to its national air quality standards, or the WHO AQG.

Instead, the categories of air quality are arbitrarily grouped to form ten equal bands throughout the entire range of %ER. If we adopt this banding system, Hong Kong might have similar proportion of days with 'low', 'moderate', 'high' and 'very high' bands as in Canadian cities, despite the obvious fact that Hong Kong's generally inferior air quality compared to Canada's. To overcome the lack of reference to any air quality standards, we have calculated the %ER based on the WHO short-term AQGs, with an adjustment to the value for NO_2, as mentioned in section 5.3. One would have expected the stringent WHO AQGs would give rise to a low additional health risk to the population. The reason for the rather high value of %ER, even at the WHO short-term AQG levels of air pollutants, is that these AQGs are themselves derived from single pollutant models. Such models attribute a measured health outcome to a single pollutant, even though the outcome is likely to be due to the combined effect of multiple pollutants. Hence, the simple summation of the %ER based on the AQGs probably overestimates the overall risk. Another reason is that we have assumed that the concentrations of all four criteria air pollutants are at the level of the AQG simultaneously, an unlikely scenario. Moreover, the calculation of %ER assumes zero risk in the absence of any air pollutants, which is hypothetical and unattainable in reality.

To further reduce the %ER to the elderly and children, who formed the most vulnerable groups to air pollution-related illnesses, we arbitrarily adjusted the % AR for the general population downwards by an adjustment factor. This reduced % AR was considered as a lower level of risk that can be tolerated by the elderly and children when air pollutant concentrations were at the WHO short-term AQG levels. We have arbitrarily designated as a health risk category of 'low', one that causes less than 50% of %ER at the WHO short-term AQG risk levels, adjusted for children. This magnitude of % AR could perhaps be regarded as a 'background risk'[§] to health posed by air pollution in an urban environment.

The health advice to the general population and the high risk age groups generally follows the Canadian approach. The only difference is in the way the health risk categories and bands are constructed, as described before. The aim is to inform and advise those at higher risk on their outdoor activities and physical exercise, including those with cardiovascular and respiratory diseases, as well as outdoor workers.[†] When the air quality is good, they can enjoy their daily outdoor activities freely; but at various poorer levels of air quality, they should be informed to take appropriate action to protect themselves from excessive exposure to outdoor air. Our method of estimating additional health risk could not distinguish between those with or without existing cardiovascular and respiratory health problems. Furthermore, it is not possible to tailor specific health advice to individual persons with health problems. Hence, people with existing cardiovascular and respiratory health problems, and people who experience chest discomfort or respiratory symptoms such as cough or breathing difficulties, must seek advice from their doctors. The general health advice given for each health risk category must be interpreted cautiously, with this limitation in mind. The purpose of sub-dividing the health risk categories into 10 bands was to provide more detailed information on the air quality and its associated health risk. We have also constructed an approximately equal increment in %ER for an increase in each band. The linearity of

[§] We should recognize the difference between this 'background' risk of air pollution and other environmental health risks, such as the background health risk of ionising radiation, which is an unavoidable risk. The 'background' risk of air pollution can be substantially reduced by concerted community efforts in modifying our way of life and reducing the use of fossil fuels.

[†] The category 'outdoor workers' was included in response to strong demand by certain sectors of the community.

the banding will facilitate the public to better understand its scale in relation to its risk, and to adjust their physical activities accordingly.

8 Conclusion and Recommendation

We have reviewed various AQI / API systems in different countries and studied their applicability to Hong Kong. We recommend that the EPD should consider using a health risk category- and band-based AQHI system that reports the health risk to the public; namely, the additional risk of hospital admissions for cardiovascular and respiratory diseases, resulting from short-term exposure to air pollutants. This short-term AQHI should be supplemented by an annual AQI for each of the criteria pollutants, anchored to the WHO AQGs, to communicate health risks from long-term exposure to air pollution.

9 Acknowledgement

This Report consists of two parts. The literature review and the comparison of API values using different levels of AQO (Chapter 3 and 4), were written by Prof. Alexis Lau and his team. The development of the AQHI (Chapters 5 and beyond) was written by Prof. TW Wong, based on statistical models developed by Prof. Wilson Tam. Prof. Tam did most of the data processing. Ms. Qiu Hong also contributed to the data analyses. Prof. Ignatius Yu and Prof. CM Wong provided input and comments. Ms. Andromeda Wong edited the Report and wrote the Instruction Manual of the AQHI Programme.

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Appendix 1: The problem of time lag for the API as an indicator of the current air pollution situation

Respirable Suspended Particulate (RSP) is a major pollutant in Hong Kong. The hourly API is calculated with reference to the past 24 hourly mean concentrations of RSP, and using the RSP 24-hr Air Quality Objective (AQO) as standard, because of the absence of a 1-hr AQO. Consequently, the hourly API often lags behind the current pollutant concentration by many hours. To illustrate this problem, the RSP concentration at Yuen Long for November 19-20, 2006 is shown below:

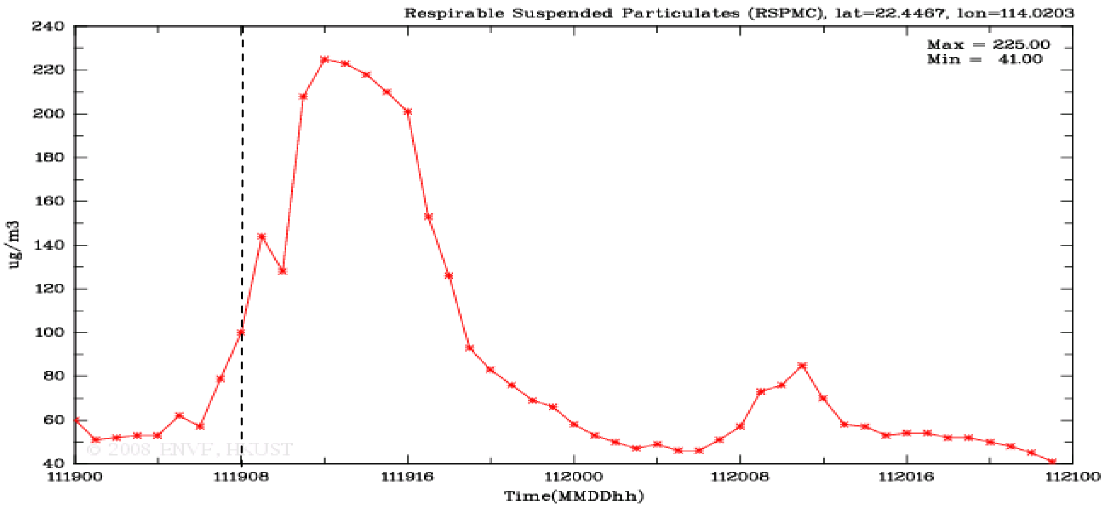

Figure 1: Hourly concentrations of respirable suspended particulates in Yuen Long, 19-20 Nov 2006

The corresponding API at the same station is shown in Figure 2. RSP was the contributing pollutant for the API. From the two plots, it is obvious that the peak in API lagged behind the peak in RSP concentrations, by approximately 12-hours.

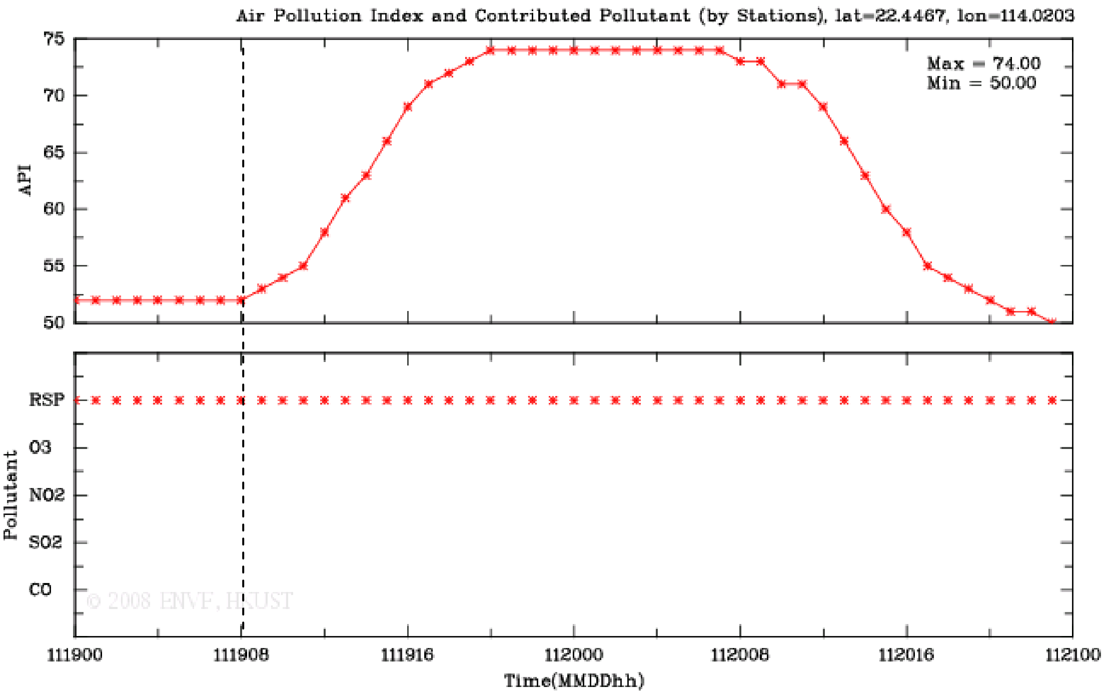

Figure 2: Hourly API in Yuen Long, 19-20 Nov 2006

A 10-km race (the 2006 Nike Hong Kong 10K challenge) was held in Tuen Mun on the morning of the 19th November 2006, which started at around 8:30 am and lasted for about 2 hours. The organizers noted that the sky was very hazy. (Visibility was down to about 4 km in Yuen Long. See Figure 3). They decided to go ahead with the race because the API at Yuen Long was only 52. Afterwards, a local newspaper†† reported that 43 people suffered from discomfort, and five participants were admitted to hospital for treatment.

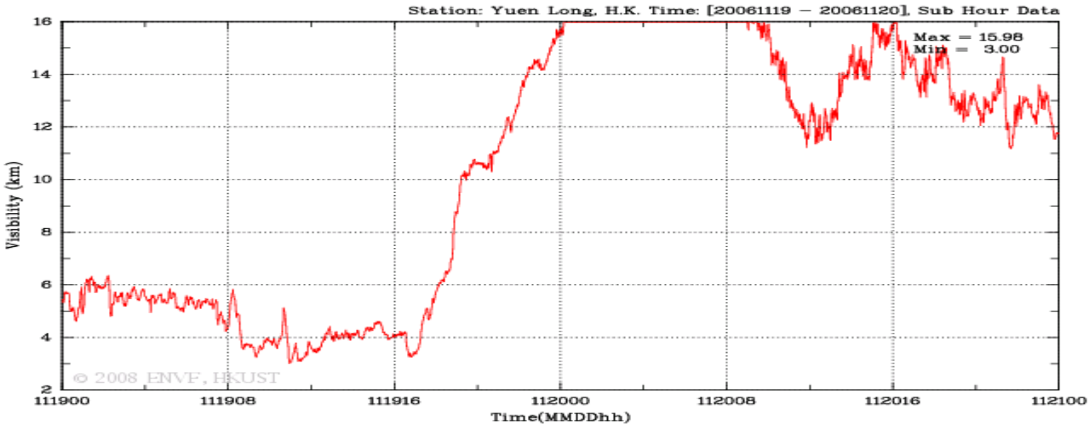

Figure 3: Visibility in Yuen Long, 19-20 Nov 2006

It is logical to assume that the organizers would postpone or cancel the race if they are aware of the rapid rise of the RSP concentrations. However, the use of the hourly API at Yuen Long (which was 52 and representative of the past 24-hour average concentration of RSP) as evidence that the air quality of Yuen Long was acceptable at the time of the race was a mistake and probably contributed to this incident. Instead, the hourly RSP concentration reported at 8am would much better information on the air quality to the organizers.

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†† Sing Tao Daily 2006-11-20

濕度高空氣濁 43 人不適 十公里賽五跑手不支送院

繼年初「渣打馬拉松」長跑，近五千名選手不適後，昨晨在天水圍濕地公園舉行的一年一度「Nike 十公里挑戰賽」，又發生同類意外，逾四十名選手疑難抵天氣潮 濕及空氣混濁，在途中相繼感到不適，其中五人更需要送院治理。主辦單位正了解事件起因，並研究日後再辦比賽時，如何避免出現同類事件。

Appendix 2: Distribution of air pollutant concentrations by AQHI bands

Table 1 Distribution of Air Pollutant Concentrations*

	NO_2	O_3	SO_2	PM_{10}
Minimum	23.33	8.42	3.93	17.51
Maximum	71.87	54.80	84.03	62.56
Range	48.53	46.38	80.10	45.04
Percentiles				
25	36.33	22.47	13.59	27.06
50	42.45	27.07	23.02	31.73
75	50.33	31.93	32.39	37.52
90	57.63	36.93	43.49	42.61

* Band = Low (1, 2, 3), Number of Days=369.

Table 2 Distribution of Air Pollutant Concentrations*

	NO_2	O_3	SO_2	PM_{10}
Minimum	38.90	8.02	4.27	22.78
Maximum	130.8	110.62	150.17	146.70
Range	91.90	102.60	145.90	123.92
Percentiles				
25	65.30	38.49	14.19	47.83
50	76.03	52.73	21.08	60.56
75	88.50	68.63	36.46	76.41
90	99.59	80.97	54.46	92.01

* Band = Moderate (4, 5, 6), Number of Days=922.

Table 3 Distribution of Air Pollutant Concentrations*

	NO_2	O_3	SO_2	PM_{10}
Minimum	64.30	16.11	7.75	51.20
Maximum	148.23	134.48	169.53	190.0
Range	83.94	118.37	161.77	138.8
Percentiles				
25	88.25	65.32	19.96	81.50
50	100.17	78.18	25.78	92.85
75	113.29	92.83	40.38	111.53
90	124.65	107.65	60.30	127.34

* Band = High (7), Number of Days=194.

Table 4 Distribution of Air Pollutant Concentrations*

	NO_2	O_3	SO_2	PM_{10}
Minimum	57.61	22.58	12.77	62.34
Maximum	207.97	171.92	211.52	227.08
Range	150.35	149.33	198.75	164.74
Percentiles				
25	107.56	72.23	29.35	102.04
50	125.51	91.17	44.92	118.41
75	142.49	116.52	67.69	135.09
90	160.47	130.58	105.35	153.3

* Band = Very high (8, 9, 10), Number of Days=292

Table 5 Distribution of Air Pollutant Concentrations*

	NO_2	O_3	SO_2	PM_{10}
Minimum	97.43	49.13	46.76	118.43
Maximum	258.66	233.31	213.47	296.06
Range	161.22	184.17	166.71	177.62
Percentiles				
25	139.40	102.53	103.03	145.33
50	163.01	163.96	132.96	168.17
75	176.65	182.43	160.89	184.96
90	214.40	196.60	182.65	213.07

* Band = Serious (10+), Number of Days=49

Appendix 3: Methodological issues in handling missing data

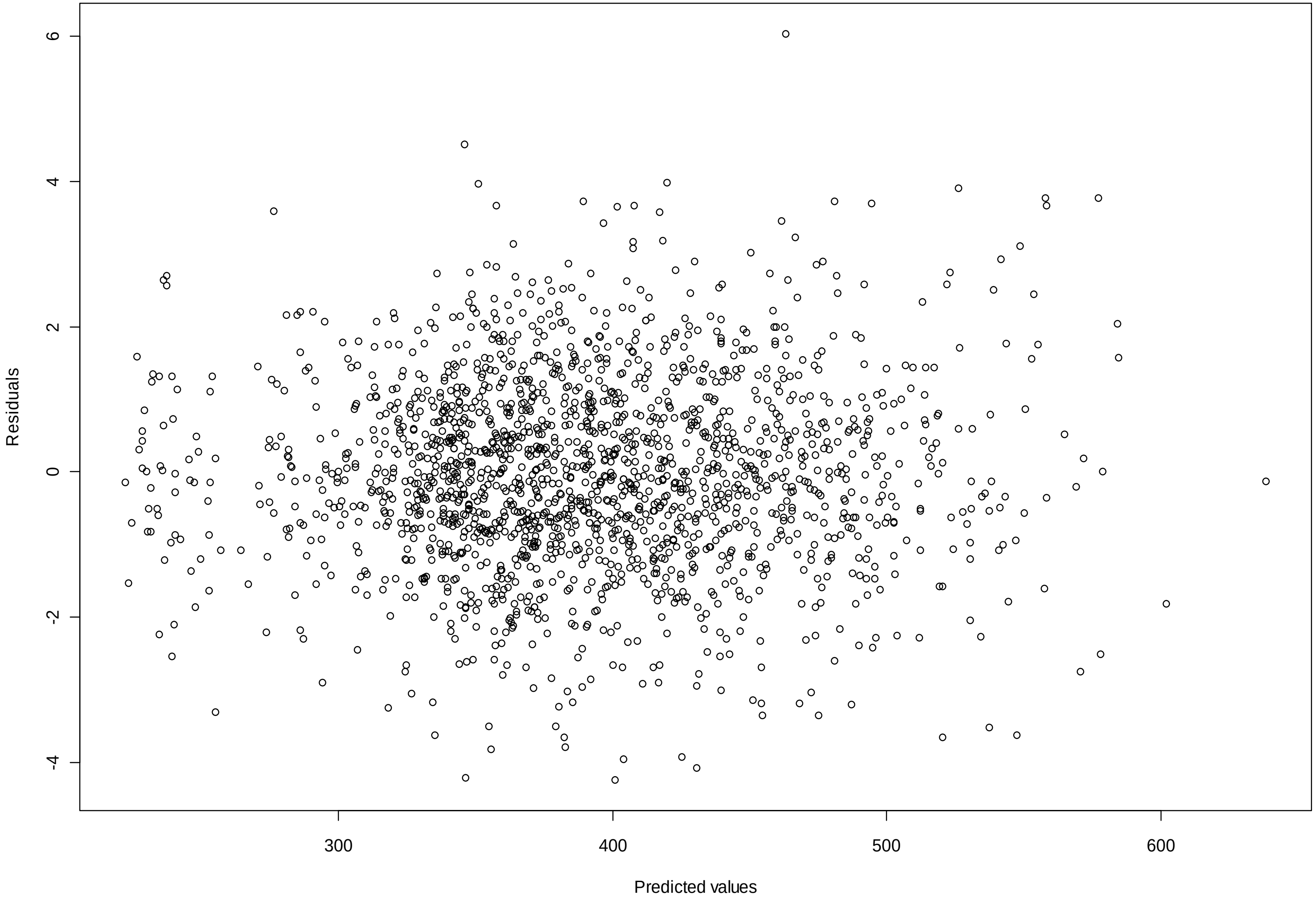
Missing data occur, albeit uncommonly, in air monitoring stations for different reasons, e.g., technical failure, machine maintenance and calibration, and other operational procedures. Since the AQHI is derived from the summation of health risks attributed to four air pollutants, the coding of unavailability of data to the null on any one will result in a lower AQHI that does not reflect the true underlying health risk. There are several ways to deal with missing data. The simplest and uncontroversial approach is not to report the district-based AQHI whenever data on one or more air pollutants are not available in a station on that day. But the AQHI in other districts with available data, and the overall, city-wide AQHI, would still be reported. However, this approach has its disadvantages. Public expectation in the reporting system is an important consideration. A high incidence of unreported AQHI would undermine public confidence.

In time series studies, it is customary to estimate missing data by imputation. A practical method that is commonly used is to impute missing data on one air pollutant in a district by using the mean concentration obtained from all the other stations with adjustment of this value by a factor reflecting the ratio of the mean of this station to the overall mean of all districts. Any missing value can then by imputed in this way in theory. However, if two out of four air pollutants are missing in a district, it is recommended that the AQHI should not be estimated.

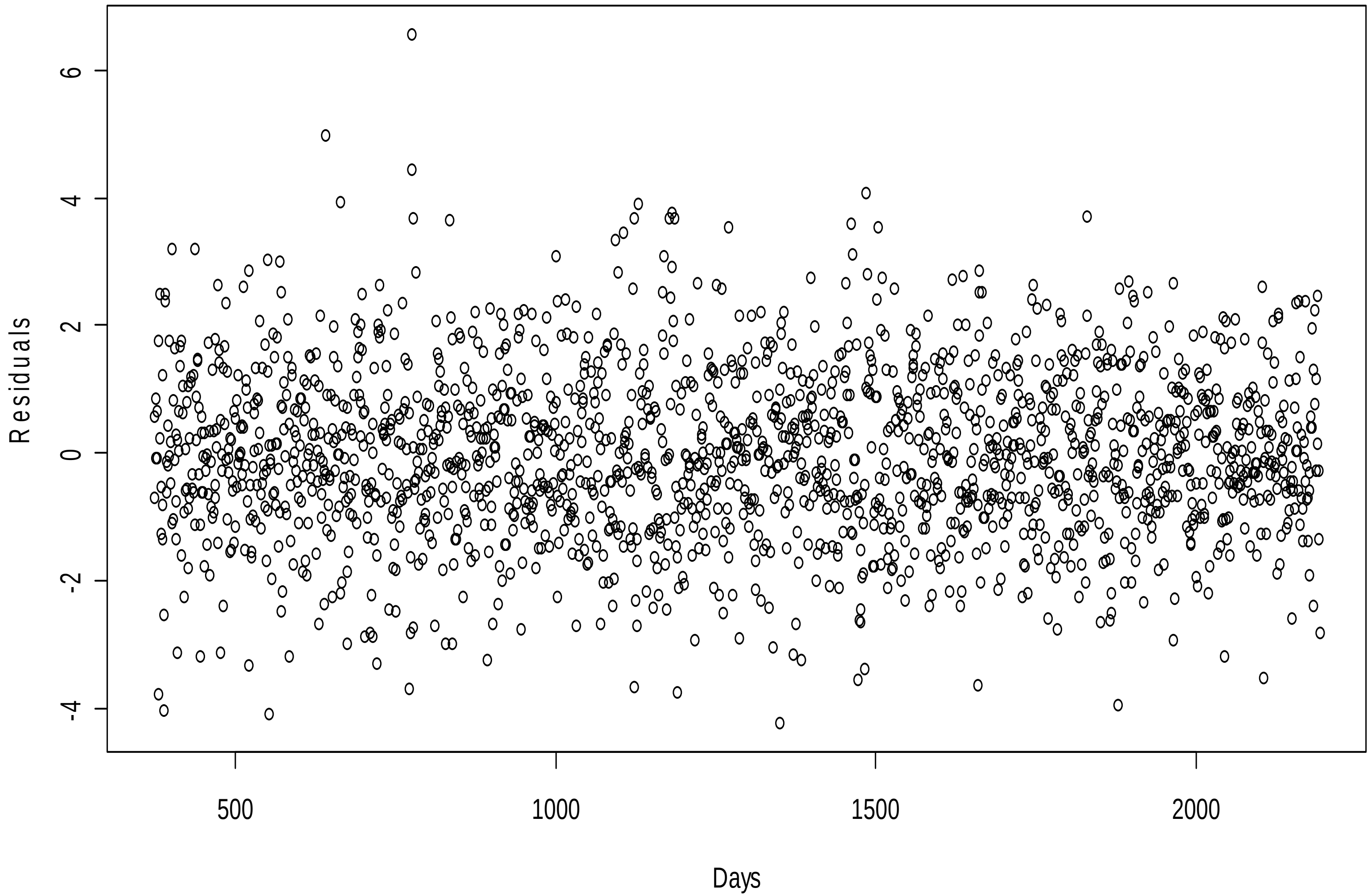
Another approach in imputing missing data is to use the data in the most adjacent station, by the regression coefficient obtained from linear regression of concentrations of that pollutant in the missing station on that in the adjacent station. The advantage of this approach is that the estimated value is probably more reliable. However, problems with this approach will arise when even the adjacent station has missing data on the same pollutant. The choice of another proxy for imputation of missing data in both stations will lead to a new problem – which adjacent station should be used? The adjustment factors of each station relative to several stations would have to be derived. Operationally, this might create confusions and errors in computation.

Irrespective of the choice of approaches (do not impute / impute with city-wide mean / impute with adjacent station), it is imperative to lay down clear and unambiguous rules that must be followed consistently.

Appendix 4: Plot of residuals against predicted no. of hospital admissions in the core model



Appendix 5: Plot of residuals in the core model (df=70) against days in the time series



Appendix 6: Partial autocorrelation function by lag days

Fig. 1: PACF plot for residuals of core model without autoregressive terms (df=70)

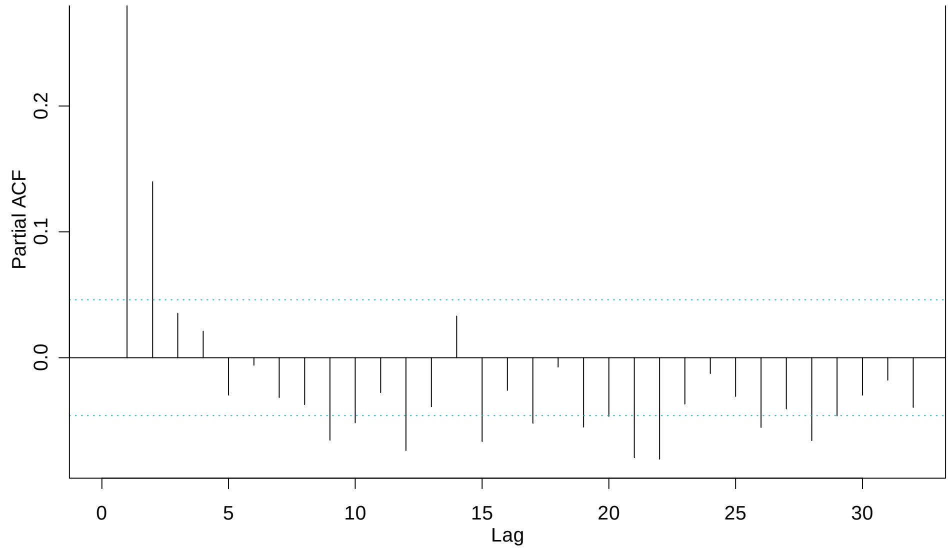
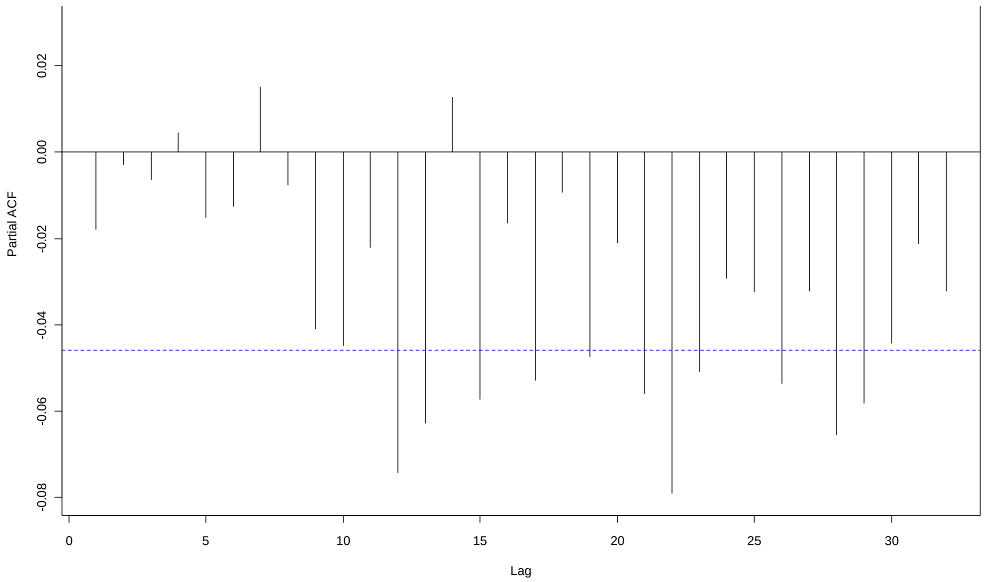


Fig. 2: PACF plot for residuals of core model with autoregressive terms included (df=70)



Appendix 7: Results of Sensitivity Analysis

The following table shows values of beta and relative risks (RR) when (i) an influenza indicator is added to the model: gam(respcardae0~respcardae1+respcardae2+respcardae3+respcardae4+respcardae5+respcardae6+respcardae7+s(day,df=70)+s(tempnew,df=15)+s(humnew,df=15)+dow+season+hind+influenza;
and when the 5 years time series (with influenza) is split into two models: 3years (2001 -03)and a 2 years (2004 – 05):

	NO_2 (lag 0)	PM_{10} (lag 0)	O_3 (lag 1)	SO_2 (lag 0)
Beta				
(i) Original model	0.0004462559	0.0002821751	0.0005116328	0.0001393235
(ii) Model + influenza	0.00043539744	0.00028084482	0.0005084575	0.00012619637
(iii) Lower 95% CI	0.0003359479	0.0001889662	0.000406226	0.00001622939
(iv) Upper 95% CI	0.000534847	0.0003727235	0.000610689	0.0002361633
(v) Difference from original*	-2.43%	-0.47%	-0.62%	-9.42%
Relative risk (per 10 µg/m^3)				
(i) Original model	1.004473	1.002826	1.005129	1.001394
(ii) Model + influenza	1.0044	1.0028	1.0051	1.0013
(iii) Lower 95% CI	1.0034	1.0019	1.0041	1.0002
(iv) Upper 95% CI	1.0054	1.0037	1.0061	1.0024
(v) Difference from original*	-0.01%	0.00%	0.00%	-0.01%
Beta				
(i) Model (2001 – 03) df=42	0.0004128346	0.0002626788	0.00059117132	0.00007448501
(ii) Lower 95% CI	0.0002746447	0.0001316061	0.000441406	-0.00009321179
(iii) Upper 95% CI	0.0005510245	0.0003937515	0.000740937	0.000242182
(iv) Difference from original*	-7.49%	-6.91%	15.55%	-46.54%
Relative risk (per 10 µg/m^3)				
(i) Model (2001 – 03) df=42	1.0041	1.0026	1.0059	1.0007
(ii) Lower 95% CI	1.0028	1.0013	1.0044	0.9991
(iii) Upper 95% CI	1.0055	1.0039	1.0074	1.0024
(iv) Difference from original*	-0.03%	-0.02%	0.08%	-0.07%
Beta				
(i) Model (2004 – 05) df=28	0.0004401971	0.0003022557	0.000425567	0.0001537015
(ii) Lower 95% CI	0.000293058	0.0001707098	0.000283149	0.000005316113
(iii) Upper 95% CI	0.000587336	0.0004338016	0.000567984	0.000302087
(iv) Difference from original*	-1.36%	7.12%	-16.82%	10.32%
Relative risk (per 10 µg/m^3)				
(i) Model (2004 – 05) df=28	1.0044	1.0030	1.0043	1.0015
(ii) Lower 95% CI	1.0029	1.0017	1.0028	1.0001
(iii) Upper 95% CI	1.0059	1.0043	1.0057	1.0030
(iv) Difference from original*	-0.01%	0.02%	-0.09%	0.01%

* Beta from original model – beta from model with influenza / beta from original model

Appendix 8: Viewpoints and discussions by team members on the Report

While all members agreed on the development of the AQHI based on the Canadian approach, several discordant opinions were raised by Prof. CM Wong on the issue of banding. **First**, Prof. CM Wong was of the opinion that the **additive approach** (which the Canadian index used) was too lax and therefore inappropriate. According to his viewpoint, the short-term WHO AQG was developed based on the assumption that an exceedence in the concentration of any one air pollutant above its AQG would cause ill-health. Therefore, for any given combination of concentrations of the 4 air pollutants, any pollutant with a concentration higher than its WHO AQG would mean a high risk to health. This is the "standard-based system" that Hong Kong has been using. This system has been criticized precisely because it focuses only on the pollutant that has exceeded the AQO, but does not take into account the effects of the other pollutants on the health of the public. To address this short-coming, the Canadian system, which our AQHI is based, uses a 'health-based' system. The additive risk approach, while not ideal, is a much more realistic indicator of overall health risk than that based on whether the concentration of one air pollutant is above the standard (AQO). This additive approach was considered by other team members (Prof. Yu and Prof. Lau) as an advantage of the AQHI over the old 'standard-based' system. Both members were supportive of the additive approach advocated by Prof. TW Wong. The **second** point raised by Prof. CM Wong was that the derivation of a **summative % excess risk** (%ERER), by adding up the %ER of the individual air pollutants **at their respective WHO short-term AQG concentrations**, might mislead the public to interpret that a summative %ER below this level ("anchor point") would imply that the concentrations of air pollutants are within the WHO short-term AQG. In fact, the %ER is an indicator of health risk, not attainment of a certain air quality standard. Although the two are related, they are by no means equivalent. The additive characteristic of the %ER is the reason for this non-equivalence. Prof. TW Wong opined, and all other team members concurred, that a clear explanation of the AQHI when it is introduced to the public is essential to avoid any misinterpretations. The view of Prof. CM Wong was that he supported the Canadian approach, but not the banding, in particular, the use of the %ER that was derived from the WHO short-term AQG. He would instead accept the Canadian approach in the banding of health risk. The Canadian system uses a 10-band scale that divides the entire range of the %ER into **10 equal bands** (1 – 10), with a 10+ for occasions that exceed the maximum %ER observed in the time series. In this "statistical" banding method, band 1-3 is labelled as good, 4-6 as fair and 7-10 as poor. All the other team members were opposed to this approach, because the arbitrary division into 10 bands depends largely on the value of the maximum %ER. Hence the cut-off points for 'good', 'fair' and 'poor' have no health basis. Instead, the use of the WHO short-term AQG to derive a %ER that serves as a boundary between 'medium risk' and 'high risk' is based on the following reasoning: In a time period when the concentrations of air pollutants are at the WHO short-term AQGs, the %ER would be at a certain level. Above this boundary level, the %ER must be considered as too high. Another disadvantage of the Canadian banding method is that it is a moving target. After a considerable improvement of air quality, the maximum %ER will fall. Yet, the proportional distribution of the 'good', 'fair' and 'poor' days will not change, if we adopt the new (lower) maximum %ER and base our banding on this value. The **third** criticism, also by Prof. CM Wong, was the choice of statistical model that we used in the derivation of the regression coefficients (β) of the air pollutants. Statistical models have been used to adjust for time-dependent variables that act as confounding factors in a time series model. There is no universally agreed "correct" model. Conventionally, a model with the "best fit", using various statistical parameters (e.g., the AIC value) is often used. We have tested the effect of model choice on the β values of the air pollutants, and found that they vary appreciably with the 'degree of freedom' (df) used in the modelling process. At the 'best fit' model (when df=147), the β will be severely biased towards ozone (O_3), which accounts for more than 50% of the %ER and particulates ($PM_{2.5}$) at 17%. While in general, the β values of the two gaseous pollutants, O_3 and nitrogen dioxide (NO_2) are larger than that for $PM_{2.5}$ and sulphur dioxide (SO_2), we found that when a df of 70 was used in modelling, the β values were much more "balanced", with O_3 contributing to 37% and PM_{10} at 20%. This choice insures that even the air pollutants with smaller β values will contribute towards the summative %ER, instead of a scenario where the %ER is overwhelmingly influenced by one dominant pollutant, O_3. We then sought to derive the maximum regression coefficients for each air pollutant, and selected a model (at df=70) where the β of all the pollutants were near their maximum values. The rationale for this arbitrary model choice was that the %ER based on a near-maximum β for each air pollutant would be a conservative estimate of health risk. This method of model choice, advocated by Prof. TW Wong, was agreed by Prof. Yu and Lau.

After much discussion, the additive approach in risk estimate, the choice of the present model, as well as the banding method, were agreed by all but one member of the team.

Appendix 9: Comments by experts in Environment Canada on the Report

The AQHI Report has been sent for comments by overseas experts and the following are comments collated by Mr. Bill Appleby Director, Environment Canada.

3.2.1 Key Air Pollutants

Just for clarification, the contributions of SO_2 and CO were not considered insignificant – let me explain the logic for the exclusion from the AQHI formulation.

SO_2 health effects in Canada could be characterized as being short lived and impactful. In the early formulation history, when SO_2 was part of the formulation, it tended to result in an over-contribution relative to its prevalence in the historical data set, i.e., Significant SO_2 concentrations were not a reoccurring geographic/spatial character of the19 year, 12 city dataset used to establish the risk coefficients for the AQHI. In addition, SO_2 concentrations have been steadily declining over the last decade.

As you point out later, the WHO (2005) recognizes the criteria air pollutants captured in the AQHI formulation are considered as "non threshold" pollutants, i.e. linear relationship between concentration and response to near 0ppb or ug/m3. There is however a recognized threshold or trigger associated with asthma of 10 minute exposure to 200ppb. The opportunity to treat SO_2 as a threshold pollutant opened up the option of using complementary messaging to the multi-pollutant health presentation of the AQHI

Upon consultation with our multi-stakeholder advisory body, it was determined that the most effective manner to handle "spiky" pollutants such as SO_2 would be to use a complementary advisory integrated into the presentation of the AQHI when thresholds were exceeded. A protocol developed for issuing an SO_2 advisory with the AQHI uses a rough equivalence of the WHO guideline (except we can not acquire 10 minute data) with the advisories issued by the local Medical Officer of Health. Often the impingement of pollution on the plume is so short in duration that by the time the advisory is issued the event has passed so the advisory issue takes into consideration the geographical risk associated with the event and the meteorology influencing its persistence.

In addition, the forecasting of the predicted multi-pollutant risk is the foundation of the AQHI. Without a trusted forecast, individuals do not have the guidance to make planning decisions to reduce their exposure to high risk pollution. The confidence levels of the Environment Canada forecasters is largely pollutant dependent with the significant confidence in the prediction of ground level ozone, and growing experience with the prediction of localized pollutants of $PM_{2.5}$ and NO_2. Prediction of SO_2, on the other hand, is a significant challenge and very hard to forecast given it is so localized and source dependent.

In the case of CO, Health Canada sensitivity analyses showed that the pollutant was highly correlated with the NO_2 which is a more robust measure of the source of combustion/traffic. Also in Canada the CO concentration measurements are so low they are close to the detection limits of the current suite of monitors in operation.

5.1 Rationale for Use of the Canadian Model

The choice of 3 hour averages for the pollutant concentration is a compromise between having stable AQHI values (to be able to issue consistent, supportive heath advice) because 1 hour data is too noisy and still having a responsive index to reflect changing meteorological conditions (moving away from long averaging periods which have characterized compliance-based standards upon which past AQI communications have been based).

5.2 Statistical Modeling

One of the premises of your analysis is that morbidity, specifically hospital admissions, is a more sensitive indicator than mortality. It is logical that one would expect that those being admitted to the hospital are less frail then those who die, and are thus perhaps more representative of a larger proportion of population. However, from a statistical point of view, in previous analyses we have found that associations with mortality were more stable and could be more precisely estimated than associations with hospital admissions. This may be because mortality is a more homogeneous outcome. In any case, I thought this was worth pointing out.

This is a fairly short term dataset (2001-2005) upon which to base the risk estimates. I am assuming that this is because the air pollution in Hong Kong is significantly greater than Canada that you are able to achieve adequate statistical power with such a short tem series.

An overall presentation of the index would be helpful (something for general public consumption) to better understand how you plan on conveying this information. I am a little confused how 3 different analyses (at risk, children and seniors) have dovetailed into an overall formulation. Are there separate AQHI for each of the populations identified?

One of the premises of your analysis is that morbidity, specifically hospital admissions, is a more sensitive indicator than mortality. I think I see where they are coming from in that one would expect that those being admitted to the hospital are less frail then those who die, and are thus perhaps more representative of a larger proportion of population. However, from a statistical point of view, in previous analyses we have found that associations with mortality were more stable and could be more precisely estimated than associations with hospital admissions. This may be because mortality is a more homogeneous outcome. In any case, I thought this was worth pointing out.

With respect to the statistical analysis, it was not entirely clear what sort of temporal smoothing function they used. Were they natural splines?

Also, there appears to be a lack of clarity around the pollutant averaging times used in the hospital admissions analysis. First impressions were that it is based on 3 hour maximum concentrations however; it appeared that in fact different averaging times were used for the different pollutants.

In terms of the scale, I understand the desire to provide a fixed point of reference such as the WHO guideline values. However, the approach that was employed is difficult to follow and results in unit increments in the index mapping to different risk increments at different points in the scale. For instance, the values 8 and 9 on the scale each span a 3 percent range of excess risk, while 7 spans only about half a percent. One result of this is that there is an apparently disproportionate frequency of days at a value of 8 on the scale compared to neighboring values. I think these discontinuities on the scale are problematic from both a scientific and a communication point of view, since they don't conform to a linear no threshold relationship. The height of the bars in figure 1 also does not seem to match the data in table 10.

It is a little unclear whether you have or have not included the PM_{10} morbidity health effects in the rollup of the risk estimates. If you have not, we would recommend that you reconsider. In the Canadian analysis, the mortality health effect associated with PM_{10} was marginally higher than that of $PM_{2.5}$ but a significant percentage of the morbidity health effect was associated with the coarse fraction. The choice to use $PM_{2.5}$ for the AQHI formulation was based on Health Canada's decision to calibrate the AQHI scale on mortality, the availability of a comprehensive air pollution data set which includes superb geographical distribution of $PM_{2.5}$ data and the focus on continued monitoring of $PM_{2.5}$. Health Canada did compute a PM_{10} risk formulation for consideration by jurisdictions which are seasonally impacted by PM_{10} (such as dust events) but few have chosen to implement and have chosen to use complementary messaging in the form of advisories to notify citizens of concern around PM_{10} driven events.

6.4 Health Risk Categories and AQHI Bands

Figure 1

It would have been nice to have some yearly or season plots to ascertain the variability of Hong Kong air quality under the AQHI banding scheme but at first glance it appears as if there would be few days in the lower risk band. With the low risk category being disproportionately smaller than those higher risk categories the preponderance of days appear to fall into the moderate and high categories. While this may be scientifically credible based on the morbidity outcomes has there been any consideration given to how this messaging would play in the pubic, i.e., message fatigue? Our experience is that a credible public health tool needs weigh the evidence of the health burden with the best practices associated with conveying "urgency "associated with high risk events. Citizens will soon "tune out" if there are too many expressed high risk events and the effectiveness of the tool with be decreased for that population.

Also, the challenge with the proposed 1-10 scale is that it is a closed scale. The Canadian experience is that to achieve an index which is sensitive to air pollution variability in "clean areas" and yet to be suitably precautionary for those communities where the air pollution may be higher risk that it is important to be able to express “extreme events”.

We do this through the use of the 10+ category which is often used for air quality events during forest fires, e.g. this summer we frequently achieved AQHI values for many small western Canadian communities in the 20+ ranges. It is a happy coincidence that these types of anomalies were rare in the data sets used to determine the risk coefficients and calibrate the index to a 1-10 scale. The result is that extreme events such as forest fires are often expressed as 10+ values and employ special advisory language to supplement the regular health messaging at this level of risk. The benefit of this approach is that these extreme events do not over-bias the day to day communications of urban air quality for the majority of Canadians. As aside, Environment Canada will also be enhancing our forecast fire prediction capacity shortly to better integrate these forecasts with the AQHI.

Health Advice – Table 1.1.

In Canada we made the decision early in message development that they should reflect “active transportation” decision making. As the WHO points out in their recent assessment of global health risk (2009), the risk from in physical inactivity (and obesity) far outweighs that attributed to urban air pollution. The health community in Canada supported the use of integrating those messages into those dealing with health advice. A recent meeting of the health practitioners, researchers and communicators in Toronto last month confirmed this and that, on the whole, the balance between promoting exercise and preventative actions is scientifically sound.

Diabetics have been indentified in recent scientific literature as being a population at risk and consideration is being given to identifying them as such associated with the Canadian AQHI health messaging.

6.5 Cold Season

The advisory committee came to a similar decision regarding arbitrarily switching to ozone-insensitive formulation over the winter season. Though more reflective of winter-time risk, the Committee rejected the proposal on the grounds of seasonal variability for application, the fact that you can get ozone in the winter and that a two formulation approach was contrary to the national, single formulation goal of the index development process.

6.6 Annual AQHI

We have been steadfast in maintaining that the index, which is based on short-term associations, should not be used as means for assessing performance of air quality regulations or a trend tool

Our experience is that there are better tools and more effective communications messaging around longer-term objectives and complications when the two are mixed. For example effective health messaging in the short term around short-term impacts is commensurate with the AQHI messaging schema (both in Canada and the proposed in Hong Kong) but more effective messages around longer-term exposure would be “don’t purchase house in the vicinity of a major roadway” or “use and lobby for mass transit". Decision to reduce long-term health effects are often more economically significant and often are a function of societal, and not, personal decision making.

Apart from mapping back to WHO single pollutant exposures, does the computation of the annual AQHI average consider long-term health effects?

Also will the annual average be calculated with air pollution data which have undergone quality assurance/quality control (QA/QC)? In Canada we have found that there are unavoidable/unforeseen data corrections which are necessary after the data have been collected and used for longer-term reporting objectives (such as compliance).

7.0 Discussion ….

The advisory committee came to a similar decision regarding arbitrarily switching to ozone-insensitive formulation over the winter season. Though more reflective of winter-time risk, the Committee rejected the proposal on the grounds of seasonal variability for application, the risk shift between seasons and that it ran contrary to the national, single formulation goal of the index development process

Your expressed confidence in being able to correlate air pollution and hospital admission is rather interesting if you were to add a forecasting component for the index. I am sure that the Hong Kong health system would appreciate a forecast of increasing human resource requirements associated with higher risk air pollution events. I did not see any reference to a predictive element anywhere in the paper.

Annex.

Missing Data

Good points.

Lets back up … the AQHI is a community health index. In those communities where there are multiple monitoring locations (large cities normally) the community value is expressed as the average of the pollutant concentrations.

The rationale for communicating a community value is based on premise that urban populations are mobile within their community and that they are exposed to different risk levels associated with pollution hourly. Community average exposures were also the basis of the analysis from which the AQHI coefficients were derived. The community value is carried on national media and on those Environment Canada's weather dissemination mechanisms.

The approach has the added benefit of increasing data availability. For those communities where there are multiple monitors, there is a significant increase in AQHI data availability because missing data are simply not considered in the computation of the average, e.g. if there are three monitoring locations within a region, and the ozone was missing for one of the monitors, the average ozone concentration used in determining the AQHI would be based on 2 and not 3 monitors.

In addition because we have chosen a 3 hour rolling average, we can calculate and publish an hourly AQHI if there is one hour of missing data. For example, when dealing with a single monitoring location with a missing hour of ozone, the concentrations from 2 of the 3 hours would be used to determine the AQHI community value. If there is more than 1 hour of missing data, the AQHI is not reported.

There is always a forecast value given for an AQHI location/region which is based on hemispheric and statistical modeling, adjacent monitoring data and the experience of the forecasters.

The central point to keep in mind is that the AQHI is an estimate of the risk associated with the air pollution mixture expressed on a scale of 1-10+. Your confidence level of communicating on that scale needs to be mirrored when developing techniques around dealing with missing data. Our experience is that where ever possible, it is desirable to avoid reporting the AQHI current conditions as unavailable.

Appendix 10: Response by the Consultants of the Hong Kong AQHI Study to Environment Canada's Comments dated 27 November 2010 on AQHI Report

The constructive comments by colleagues in Environment Canada on our proposed AQHI system are much appreciated. In addition to our responses, we would also like to take this opportunity to elaborate the features of our proposed system including the rationale of our choices of models and decisions on the banding in the development of our system.

1. On the choice of air pollutants

Any reporting system adopted by a community has to take into account the characteristics of that community, in particular the profile of air pollutants. In Hong Kong, sulphur dioxide (SO_2) is much less a problem since the early 1990s, when stringent measures to restrict the sulphur content of diesel fuel were implemented. We suspect that the relatively low concentrations of SO_2 compared to the other pollutants – NO_2, PM and O_3 account for our findings in the time series analysis of a much lower (though statistically significant) relative risk for SO_2 than the other 3 pollutants. Resulting from this, the contribution of SO_2 to the overall %excess risk (%ERER) is small compared to the other pollutants, in particular, NO_2 and O_3, whose RRs as well as concentrations are much higher. Nevertheless, we have decided to include SO_2 in the overall calculation of the%ER, to cater for situations when the concentrations of SO_2 might increase due to meteorological or other external factors.

Similar to Canada, we found that carbon monoxide has not been a major problem in Hong Kong.

2. On the prediction features of AQHI

We agree with Environment Canada's view that the ability to accurately forecast the levels of air pollutants is very important to the proper functioning of the AQHI. The prediction of air pollutant concentrations remains a challenging task for the Hong Kong Environmental Protection Department (EPD).

3. On modelling issues

Our choice of the 3-hour averaging time was strictly an adoption of the Canadian system and not based on sensitivity tests of different averaging times. We are aware of the fine balance between one with a reasonably short lag time and one that fluctuates too widely to be stable. As for our decision to use emergency hospital admissions for cardiovascular and respiratory illnesses in our time series analysis, we have acquired much experience in past epidemiological studies using this dataset, which, as we mentioned in the Report, is comprehensive in coverage (over 90% of hospital beds and an even higher % of emergency hospital admissions), uniform (using one computerized health information system since the mid-1990s) and credible (a quality control system is in place). Another reason for our choice is that, as pointed out by Environment Canada, hospital admissions represents a broader band of the 'pyramid' of health outcomes than mortality. Our risk estimates are based on a 3-hour averaging time for air pollutant concentrations. Although the magnitude of the RRs was different from that found our previous work using daily averages of air pollutant concentrations, they are similar in terms of their respective statistical significance. The length of our hospital dataset was 5 years. Further analyses using a longer time series should be done to test the stability of the risk estimates.

In response to Environment Canada's queries about modelling issues, we would like to explain in more details about our approach. In the model construction, we used a generalized additive model and smoothing splines for the adjustment of time-dependent confounders, using the S plus software. As mentioned earlier, we used the maximum of the 3-hr moving averages for each day, identical to that used in the Canadian method. As for the RR of different lag days for different pollutants, we chose the lag day that gives the maximum RR.]

There is no consensus on the degree of freedom used in such modelling, and we found that if we used the model with the minimum AIC (one of the conventional approaches in time series studies) the result would seriously bias the summed %ER towards O_3, the pollutant with the highest RR (It accounted for over 50% of the summed %ER). We instead chose a model with a degree of freedom that gave a maximal / near maximal RR for most pollutants except SO_2 (whose RR is much lower than the others for any degree of freedom). This choice results in a more even contribution of %ER by 3 of the 4 pollutants – SO_2 is still unimportant; O_3 still has the highest RR, followed closely by NO_2, while the RR of PM is a bit lower. This pattern of RR (higher for NO_2 and O_3 than for PM) agrees with our previous work that used daily mean air pollutant concentrations in the time series analysis with hospital admissions as health outcomes. Moreover, the use of a near-maximal RR ensures that we would prefer to be conservative in our risk reporting rather than under-estimate the health risk.

4. On the presentation to the public and the RR for children and the elderly

Before the index is launched, we would have presentations to different groups – professionals, environmental groups and the general public. There is only one AQHI. The RRs derived for the elderly population and those aged <5 years were simply used as adjustment factors to obtain an %ER at which value the health risk is considered to be high for the two groups, taking the lower %ER of the 2 groups. In brief, the %ER obtained for the entire population is adjusted by an adjustment factor derived from a ratio of the median %ER of the children or elderly, whichever the higher, to the median %ER for the general population. Only one set of data, based on the RR for the general population was used. The %ER derived from the adjustment factor was used to demarcate the "high health risk" band (i.e., risky to susceptible groups) from the "very high risk" band (i.e., risky to the general population).

5. On comments about the unequal increments in the AQHI scale and possible errors in Figure 1

The unequal increment in the scale is related to our method of banding that is linked to the %ER applicable to the general population and also a %ER that is devised for the susceptible groups. We are not too worried about this feature from a scientific point of view, because we can explain why and how we derived different %ER for the different groups. Basically, the underlying concept is that we consider that to better protect the susceptible groups, we would adjust the risk level downwards by a factor obtained from %ER calculations based on a higher RR, separately derived from models for the two susceptible age groups. We think that as long as we are clear and open with our basic assumptions and methods, the idea should be quite acceptable to the public. We have checked for errors in Figure 1. The table and graphs are actually in agreement. The height of the bars does not represent the bands. Instead, they are different values of %ER.

6. PM_{10} versus $PM_{2.5}$

We have more comprehensive dataset with PM_{10}. The general consensus is that $PM_{2.5}$, being much smaller in size, is more harmful to health than PM_{10}. This is the main reason for our EPD to increase the number of monitoring sites for $PM_{2.5}$. Instead, we chose to use PM_{10} as a component to calculate the %ER instead of $PM_{2.5}$. The reason for our choice is that data on PM10 are more comprehensive in all air monitoring stations, whereas that on PM2.5 were limited to a few stations only. Hence the RR derived for PM_{10} is considered more robust than that derived from $PM_{2.5}$. Our RR for PM_{10} is slightly larger than that for $PM_{2.5}$, similar to findings in Canada. This indicates that the smaller, more penetrating $PM_{2.5}$ have instead a smaller short-term effect on health than PM_{10} do. The concentration of PM_{10} is higher than that of $PM_{2.5}$. This would give a somewhat higher % contribution of PM in the %ER. We are aware of the problem of coarse particles, which could reach very high concentrations during dust storms from northern China. These events can be readily identified by our AQHI, which is based on PM_{10}.

7. On the issue of "message fatigue"

Message fatigue is a major issue in risk communication. After all, the no-threshold concept of PM and possibly O3 would mean there always exists some health risk. We cannot ask the public to stay home whenever a health risk exists from outdoor air pollution. There has to be a balance between risk-taking and restriction of outdoor physical activities, as Environment Canada rightly pointed out about the substantial health risk of physical inactivity. We are aware that we have fewer days in the 'low risk' band. One reason is that Hong Kong has worse air quality than most cities in Canada. We are also wary of possible scepticism by the public that the EPD is deliberately toning down the air pollution problem through the public announcement of many 'low risk' days. We need to do a skillful job in risk communication to the public and introduce this AQHI system accurately to avoid a perception that our EPD is down-playing the health risk. As for the suggestion of an additional 10+ band in the scale, our 10th band is open-ended that includes all the %ER above 19.37%. This

%ER is just below the 98^{th} percentile of the %ER in the 5-year dataset.

8. On the issue of the long-term AQI

We agree with Environment Canada's view that the AQHI is a short-term risk communication system, and not linked to long-term policy decisions. To convey this message, the public's expectation of the API needs to be changed, through better communication and explanation of the AQHI system, as well as better explanation of air pollution control policies. The annual AQI is intended to reflect long-term health risk, using the WHO annual AQG as a standard. The data used will have been quality-checked before being used. This annual AQI is not designed to be a communication tool of the day-to-day health risk.

9. On the issue of forecasting of the AQHI

It is not the objective of the AQHI to forecast hospital bed use. We need to do much more to achieve this goal. For example, studies in the United States have suggested that temperature changes, such as cold spells and heat waves, are strong predictors of hospital bed use. This would be a separate study, probably a joint study with our Health Bureau or Hospital Authority.

Professor TW Wong
Principal Consultant to the AQHI Study
23 May 2011

	no2	rsp	o3	so2	fsp
Daily	7.80E-04	4.03E-04	1.02E-03	2.45E-04	4.44E-04
max-3 hour	4.46E-04	2.82E-04	5.12E-04	1.39E-04	2.18E-04
%	57.22%	70.06%	50.11%	56.85%	49.11%

% refers to the beta value derived from max-3 hr pollutatn conc. / beta value derived from daily pollutant conc. expressed as a %

Additional analyses by Wilson: beta derived from 3-hr max air pollutant concentrations in the regression model as a % of beta derived from using 24-hr mean air pollutant concentration in the model for the 4 pollutants are shown in the above table. Note that all values are smaller using the 3-hr max, compared to using the 24-hr mean. This is similar to Stieb's findings (2008) in the JAWM journal (the long paper). We cannot compare the beta values (or RRs) from our study with that in Stieb's study, because he used mortality while we used hospital admissions as health outcomes.