Available online at www.sciencedirect.com
ScienceDirect
Atmospheric Environment 41 (2007) 8442–8454
ATMOSPHERIC ENVIRONMENT
www.elsevier.com/locate/atmosenv

A novel air pollution index based on the relative risk of daily mortality associated with short-term exposure to common air pollutants

Eugene K. Cairncross[a,*], Juanette John[b], Mark Zunckel[c]

[a]*Department of Chemical Engineering, Cape Peninsula University of Technology, Box 1906, Symphony Way, Bellville 7535, Western Cape, South Africa*
[b]*CSIR, P.O. Box 395, Pretoria 0001, South Africa*
[c]*CSIR, P.O. Box 17001, Congella 4013, Kwa-Zulu Natal, South Africa*

Received 28 November 2006; received in revised form 28 June 2007; accepted 3 July 2007

Abstract

Communication of the complex relationship between air pollutant exposure and ill health is essential to an air pollution information system. We propose a novel air pollution index (API) system based on the relative risk of the well-established increased daily mortality associated with short-term exposure to common air pollutants: particulate matter (PM_{10}, $PM_{2.5}$), sulphur dioxide, ozone, nitrogen dioxide and carbon monoxide.

To construct our index system, the total incremental daily mortality risk of exposure to these pollutants was associated with an index value ranging from 0 to 10. The index scale is linear with respect to incremental risk. The index is open ended, although, for convenience, an index of 10 is assigned for exposures yielding indices $\geqslant 10$.

To illustrate the application of this API system, a set of published relative risk factors are used to calculate sub-index values for each pollutant, in the range of air pollutant concentrations commonly experienced in urban areas. To account for the reality of ubiquitous simultaneous exposure to a mixture of the common air pollutants, the final API is the sum of the normalised values of the individual indices for PM_{10}, $PM_{2.5}$, sulphur dioxide, ozone, nitrogen dioxide and carbon monoxide. This establishes a self-consistent index system where a given index value corresponds to the same daily mortality risk associated with the combined exposure to the common air pollutants. To facilitate health-risk communication, index values are colour coded and associated with broad health-risk descriptors. The utility of the proposed API is illustrated by applying it to monitored ambient concentration data for the City of Cape Town, South Africa.

Keywords: Air pollution/quality index; Health-risk communication; DAPPS; Multiple pollutant exposure; South Africa

1. Introduction

Urban ambient air pollution is the result of emissions from a multiplicity of sources, mainly stationary, industrial and domestic fossil fuel combustion, and petrol and diesel vehicle emissions (Brulfert et al., 2005; Parra et al., 2006). Ambient

*Corresponding author. Tel.: +27 21 959 6083/6490, +27 82 200 7056; fax: +27 21 959 6083.
E-mail addresses: cairncrosse@cput.ac.za (E.K. Cairncross), jjohn@csir.co.za (J. John), mzunckel@csir.co.za (M. Zunckel).

1352-2310/$ - see front matter
doi:10.1016/j.atmosenv.2007.07.003

pollutant concentrations are the resultant of primary pollutant emissions from these sources, atmospheric transformation processes, including the formation of secondary pollutants, and dispersion influenced by local topographical features and meteorological conditions (Turner, 1994; Singh, 1995). These heterogeneous pollutant sources and processes result in pollutant concentrations that vary with time and location within the urban environment, independently (if, for example, emitted from different sources), collinearly (if emitted from the same source and subject to similar atmospheric behaviour) or antagonistically (the titration of O_3 against NO) to each other. The inhabitants of a typical urban centre may be exposed to about 40 individual chemicals and/or groups of chemicals, totalling more than 100 individual chemical species (WHO, 2000a).

The adverse health effects associated with air pollution may be attributable to short-term (a few minutes to 24 h) exposure or long-term (months to years to decades) exposure, and different pollutants may have widely different exposure–response characteristics. An extensive literature (reviewed, for example, in WHO, 2000a, b, 2001a, 2005; Maynard, 2004; Brunekreef and Holgate, 2002; Curtis et al., 2006) has demonstrated the associations between exposure to the classical pollutants and ill-health endpoints such as increased hospital admissions for respiratory, cardiovascular disease and congestive heart failure, increases in asthma attacks, increases in acute bronchitis and decreased lung function. Many studies have also shown the association with increased daily mortality, in total and due to cardiovascular and respiratory causes. Short-term adverse health responses may occur within minutes—for example, asthmatics exposed to SO_2 may experience effects within minutes (WHO, 2000b)—or may lag the exposure by several hours, up to a period of several days (Lipfert, 1994). The long-term health effects of exposure to particulate matter (PM) are associated with shortening of life expectancy, increased rates of bronchitis and reduced lung function; the separate effects of long-term exposure to SO_2, NO_2, O_3 and CO are less certain, but studies have demonstrated, variously, associations with decreased lung function, increased bronchitis symptoms and increased prevalence as well as exacerbation of asthma (Brunekreef and Holgate, 2002; Maynard, 2004; Kyle et al., 2002). Carcinogens such as benzene have a latency period of years or decades (WHO, 2000a). Further factors that complicate attempts to accurately estimate the actual site-specific health risks associated with air pollution include differences in individual exposure and susceptibility, the local prevalence of health conditions that may predispose the exposed population to an adverse response as well as the ability of the population to recover from or cope with these exposures (US EPA, 2003; Kasperson and Kasperson, 2001).

Worldwide, many cities continuously assess air quality using monitoring networks designed to measure and record air pollution concentrations at several points deemed to represent exposure of the population to these pollutants. The purpose of such a network is several fold—to compare measured values against guidelines or standards, to assess the success or otherwise of pollution reduction strategies and to monitor medium and long-term trends, among others. Since air quality guidelines or standards are usually based on considerations of the likely adverse health impact of pollutant levels, the comparison of measured values against guidelines (or standards) implicitly conveys the message that the air quality is acceptable, from a health perspective, or not, depending on whether or not the guidelines are exceeded. The output of a monitoring network, i.e. pollutant concentrations, may be conveyed to the public through periodic reports that include concentration time series, and a comparison of the measured values for each pollutant against the applicable guideline. Current research indicates that guideline values cannot be regarded as threshold values below which a zero adverse response may be expected (WHO, 2000a; Koenig and Mar, 2000; Gent et al., 2003). Therefore, the simplistic comparison of measured values against guidelines may be misleading unless appropriately qualified. A more sophisticated and widely used approach is to communicate the health risk of ambient concentrations by using an air pollution (or air quality, AQI) index (API). This is typically a numerical scale, usually colour coded, intended to convey the likely severity of the adverse health effects at the monitored concentration levels (Maynard and Coster, 1999).

The communication of the complex relationship between air pollutant exposure and ill health in a manner that is both simple and accurate is thus an important albeit difficult aspect of an air pollution information system. Such health-risk communication may have several objectives—to enable the public to understand the likely health and environmental impacts of air pollution, to encourage a reduction in activities that contribute to air pollution, to enable sensitive groups such as asthmatics to take precautionary measures, to enable the public to assess pollution trends and to increase awareness of the public health implications of air pollution (Maynard and Coster, 1999; Stieb et al., 2005). Payne-Sturges et al. (2004) argued that a risk-based approach to communicating ambient exposures to a community enables the simultaneous consideration of pollutant toxicity, provides a common denominator for the comparison of risks and the setting of priorities and, by summing risks, communicates information regarding cumulative exposures. They showed that a risk-based approach to presenting exposure results provides a means to move beyond traditional reporting of concentration values.

In practice a subset of air pollutant exposures, consisting of the concentrations of the common air pollutants—sulphur dioxide (SO_2), PM, nitrogen oxides (NO_x), carbon monoxide (CO) and ozone (O_3)—is used to determine air pollution (or quality) indices. Measured or modelled ambient concentrations may be used as surrogates of exposure. To promote public access to the information, the index values for a given city or region are usually published on the World Wide Web.

In spite of the widespread use of air pollution (quality) index systems, there is currently no internationally accepted methodology for constructing such a system (Maynard and Coster, 1999; Stieb et al., 2005). Four key problems have to be addressed in the design of an API system. These are the selection of pollutant exposure metrics (pollutants and corresponding concentration averaging times) for inclusion in the index, the selection of appropriate exposure (health)–response functions, the choice of a relative scale—the exposure metric value that corresponds to an index value—and a methodology or algorithm for calculating the overall index value (the ultimate objective) for simultaneous exposure to a number of pollutants. In addition, appropriate descriptors of the risk levels corresponding to index values are required to facilitate communication of the associated health risk.

We propose an approach that systematically addresses the above key requirements of an API system. The proposed index system is based on the relative risk of premature daily mortality due to simultaneous exposure to the five common air pollutants. The application of our API is illustrated using a set of published relative risk factors and monitored ambient concentration data for the City of Cape Town, South Africa.

2. Current API and AQI systems

A number of countries and territories (including the United Kingdom (UK), the United States of America (USA), Belgium, France, Spain, Finland, Sweden, Canada, Mexico, Australia, New Zealand, Hong Kong, Singapore, Malaysia, Thailand, China, Macau, Indonesia, Taiwan) use an API (or AQI), usually applied at the urban (city) scale, to communicate air quality. In the majority of examples, the API is based on the ambient concentrations of common pollutants—SO_2, PM_{10}, NO_2, CO and O_3. In a few cases $PM_{2.5}$ is considered in the calculation of the index. Most of the systems surveyed use measured (monitored) data rather than modelled air pollutant concentrations to assess population exposure. Current international practice is illustrated through a more detailed discussion of the UK and USA systems.

2.1. The UK index system

The UK API system (Table 1) was originally introduced in 1990 as a four-band system indicating low, moderate, high and very high air pollution levels. In 1997 this system was modified to a 1–10 index scale by breaking each of the low, moderate and high bands into three equal index values (i.e. 1–9) with values greater than the high/very high threshold being designated index 10 (Maynard and Coster, 1999). The breakpoint value between the 'low' and 'moderate' bands (index values 3–4) corresponds to the UK Air Quality Standards; the air quality standards are based on the assessment of adverse health effects of air pollution.

The rationale behind this index system is given as follows (UK National Air Quality Information Archive; Joseph, 2002):

LOW (1–3): Effects are unlikely to be noticed even by those sensitive to air pollution.
MODERATE (4–6): Sensitive people may notice mild effects but these are unlikely to need action.
HIGH (7–9): Sensitive people may notice significant effects and may need to take action.
VERY HIGH (10): Effects on sensitive people, described for HIGH pollution, may worsen.

Table 1
Boundary values between index points for each pollutant in UK system (adapted from NETCEN, 2006)

Band	Index	Ozone[a] Eight-hourly or hourly mean ($\mu g\,m^{-3}$)	Nitrogen Hourly mean ($\mu g\,m^{-3}$)	Sulphur dioxide 15 min mean ($\mu g\,m^{-3}$)	Carbon dioxide 8-h mean ($mg\,m^{-3}$)	PM_{10} monoxide particles 24-h mean ($\mu g\,m^{-3}$)
Low	1	0–32	0–95	0–88	0–3.8	0–16
	2	33–66	96–190	89–176	3.9–7.6	17–32
	3	67–99	191–286	177–265	7.7–11.5	33–49
UK Air Quality Standards						
Moderate	4	100–126	287–381	266–354	11.6–13.4	50–57
	5	127–152	382–476	355–442	13.5–15.4	58–66
	6	153–179	478–572	443–531	15.5–17.3	67–74
High	7	180–239	573–635	532–708	17.4–19.2	75–82
	8	240–299	636–700	709–886	19.3–21.2	83–91
	9	300–359	701–763	887–1063	21.3–23.1	92–99
Very high	10	360 or more	764 or more	1064 or more	23.2 or more	100 or more

[a]For O_3, the maximum of the 8-hourly and hourly mean is used to calculate the index value.

The index values in the range 0–9 are approximately linear with respect to pollutant concentrations. For each pollutant exposure metric, the lower bound of index value 4 equals the UK Air Quality Standards, as recommended by the Expert Panel on Air Quality Standards (Maynard and Coster, 1999). The standards are in effect the basis for normalising (providing a relative scale) the index system.

2.2. The US Environmental Protection Agency (US EPA) system

An AQI, originally called the pollutant standard index, was established in 1976, for use by state and local agencies on a voluntary basis (Table 2) (US EPA, 1998).

The AQI includes indices for O_3, PM, CO, SO_2 and NO_2. For each pollutant, ambient concentrations are related to index values on a scale from 0 to 500, representing a very broad range of air quality, from pristine air to pollution levels that present an imminent and substantial endangerment to the public (US EPA, 1999). The index is normalised by defining an index value of 100 as that corresponding to the primary National Ambient Air Quality Standard (NAAQS) for each pollutant, and an index value of 500 as the 'significant harm level' (SHL). Such index values serve to divide the index into categories, with each category being identified by a simple informative descriptor. The descriptors are intended to convey information about how air quality within each category relates to public health, with increasing public health concerns being conveyed as the categories approach the upper end of the scale (Lipfert, 1994).

For pollutant concentrations within the various bands, the US EPA assumes that the exposure–response functions are linear within the bands, and provides a linear interpolation procedure to estimate index values between the breakpoints.

The index values, descriptors and colours associated with the US EPA AQI system are:

0–50: Conveys a positive message about air quality.
51–100: Conveys a message that daily air quality is acceptable from public health perspective, but every day in this range could result in potential for chronic health effect; and for O_3, convey a limited health notice for extremely sensitive individuals.
101–150: Conveys a health message for members of sensitive groups.
150–200: Requires a health advisory of more serious effects for sensitive groups and notice of possible effects for general population when appropriate.
201–300: Health alert of more serious effects for sensitive groups and the general population.
301–500: Health warnings of emergency conditions.

Table 2
Breakpoints for USA air quality index (adapted from WHO, 2001a; Lipfert, 1994)

These breakpoints							Equal these AQIs	Categories
O_3 ($\mu g\,m^{-3}$) 8 h	O_3 ($\mu g\,m^{-3}$) 1 h[a]	PM_{10} ($\mu g\,m^{-3}$) 24 h	$PM_{2.5}$ ($\mu g\,m^{-3}$) 24 h	CO ($mg\,m^{-3}$) 8 h	SO_2 ($\mu g\,m^{-3}$) 24 h	NO_2 ($\mu g\,m^{-3}$) 24 h		
0–28		0–54	0–5.4	0–5.1	0–90	[b]	0–0	Good
130–168		55–154	15.5–40	5.2–10.9	93–383	[b]	51–00	Moderate
US National Air Quality Standards								
170–208	250–328	155–254	41–65	11.0–14.4	386–596	[b]	101–50	Unhealthy for sensitive groups
210–248	330–408	255–354	66–150[c]	14.5–17.9	599–809	[b]	151–200	Unhealthy
250–748	410–808	355–424	151–250[c]	18.0–35.3	811–1607	124–237	201–300	Very unhealthy
[d]	10–1008	425–504	251–350[c]	35.4–46.9	1609–2139	239–313	301–400	Hazardous
[d]	1010–1208	505–604	351–500[c]	47.0–58.5	2141–2671	315–390	401–500	Hazardous

[a]The AQI report may be based on 8-h O_3 values. In some cases the 1-h O_3 index value may be calculated and the maximum of the two reported.
[b]NO_2 has no short-term NAAQS and can generate an AQI only above an AQI value of 200.
[c]If a different SHL (significant harm level) for $PM_{2.5}$ is promulgated (in the US), these numbers will change accordingly.
[d]Eight-hour O_3 values do not define higher AQI values ($\geqslant$301). AQI values of 301 or higher are calculated with 1-h O_3 concentrations.

2.3. Comparison of the UK and USA index systems

Although the UK and USA API systems attempt to achieve the same objective, i.e. the presentation of air pollution data using an index system, they differ in several significant respects. These are:

- The UK index system has values from 0 to 10, with 10 (designated a 'very high' pollution level) representing all concentrations greater than the upper bound for the eight to nine band; the US system has values from 0 to 500, values in the range 300–500 are designated 'hazardous'.
- Ozone 1- and 8-h averages are used in both cases, but these values are used somewhat differently. In the US system, both 1- and 8-h averages O_3 concentration values are used to define index values in the range 101–300, but in the UK system, either the 1- or the 8-h concentration value may be used to define the index value. The air quality standard for O_3 is $100\,\mu g\,m^{-3}$ for the 1- *or* the 8-h average value; the US standard is $250\,\mu g\,m^{-3}$ for the 1-h average and $170\,\mu g\,m^{-3}$ for the 8-h average.
- In the cases of SO_2 and NO_2, different time-averaged values are used. The US EPA includes $PM_{2.5}$ in its index system whereas the UK does not.
- The UK and US Air Quality Standards for CO are essentially the same, 11.6 and $11.0\,mg\,m^{-3}$, respectively. In the UK system the AQS value for 24-h average PM_{10} is $50\,\mu g\,m^{-3}$; in the US system it is $155\,\mu g\,m^{-3}$, about 3 times higher than the UK value.
- The breakpoints between the 'low' and 'moderate' bands (between index values 3 and 4) in the UK system for PM_{10} is $50\,\mu g\,m^{-3}$; the US value between 'good' and 'moderate' (US index value 100) is similar—$54\,\mu g\,m^{-3}$. For CO, the corresponding values are 11.6 ppm (UK) and 11.0 ppm (US). Thus for PM_{10} and CO the descriptors are reasonably aligned. However, for O_3, the UK 'low' to moderate' breakpoint is 49–50 ppb (8-h average); for the US the corresponding breakpoint between 'good' and moderate is somewhat higher, at 64–65 ppb. The other pollutants cannot be directly compared because different averaging periods are used in the two countries. The descriptors for similar exposures differ significantly. Thus, both the index values and the more general descriptors of 'low', moderate', 'high' and 'very high' in the UK cannot readily be aligned with the US descriptors of 'good', 'moderate', etc.

In several countries, including the UK and the USA, the descriptor of the air quality for the day is taken as the highest reached by any pollutant of the group that is monitored. If only one pollutant reaches the 'moderate band' levels of air pollution, the descriptor used is 'moderate'. If, for example, four pollutants all reach the moderate band air pollution, it is again described as moderate. However, in the second case, a more significant health effect may be expected in comparison to the former (Maynard and Coster, 1999).

2.4. API systems

The literature on the underlying basis of API systems is comparatively sparse. Swamee and Tyagi (1999) proposed an equation that yields an 'aggregate API' deemed to account for multiple pollutant exposure. However, the functional form of the equation and weighting factors used, whilst satisfying certain mathematical criteria, are essentially arbitrary, without reference to the epidemiology of the pollutants included in the analysis. Khanna (2000) proposed an index of air pollution based on the United States' pollutant standards index and the US NAAQS embedded in the US index. Khanna (2000) used a non-linear 'damage function' relating 'welfare losses' to pollutant concentrations as a common metric to aggregate the impact of the pollutants included in the US pollutant standard index. A heuristic argument, based on micro-economic theory, is used to suggest the functional form of the 'welfare loss' function. Khanna's analysis is limited by the assumption of a hypothetical 'welfare loss' function that includes a discontinuity at an arbitrary maximum pollutant concentration level; to calculate the overall index, pollutant concentrations are equally weighted with respect to damage. Sharma et al. (2003) proposed an AQI system for India that is essentially an adaptation of the US system. Similarly, Trozzi et al. (1999) proposed a system for Italy based on that of the US.

Kyle et al. (2002) addressed the question of the relationship between health effects attributable to short-term exposure and those attributable to long-term exposure. They proposed an aggregate index that represents the adverse health effects of long-term exposure to the five common air pollutants (CO, NO_2, O_3, PM_{10} and SO_2). Their proposed aggregate index value is the sum of index ratios (the ratio of monitored values to the applicable US NAAQS) of each of the pollutants, converted to a 100-point scale where 100 would represent the long-term pollution equal to the five standards for all five pollutants.

Stieb et al. (2005) used an extensive daily time-series study to develop a no-threshold, multi-pollutant AQI based on the relationship between CO, NO_2, SO_2 and $PM_{2.5}$ ambient concentrations and mortality in Canadian cities. The derived risk coefficients were applied to daily air pollution concentrations to calculate multi-pollutant percent excess mortality, and the results were scaled from 0 to 10, with the value of 10 corresponding to the highest observed value in an initial data set. The method was applied to monitored concentrations in seven Canadian cities.

3. Methodology

3.1. Methodology for developing the API system

Modelled or monitored pollutant concentrations and published exposure–response relative risk functions for a given health endpoint are used to derive a numerical scale specific to each of the pollutants to be included in the index system. The factors considered in constructing the API system are:

- The pollutants and their ambient concentration averaging period(s) (the surrogate exposure metric) to be considered for each pollutant.
- The health endpoints and response time of exposure to the air pollutants, the availability of exposure–response relationships for each exposure metric in relation to each health endpoint, including a consideration of the 'toxicological model' (Lipfert, 1994) for exposure–response to be used.
- The relative scale (the basis for normalising the data) to be used as a normalised numerical scale to establish an equivalence of harm for different pollutants, that may have different health endpoints and have different exposure–response relationships.

Specific criteria used to screen the pollutant exposure metrics to be included in the API system are an adverse health response time of <3 days, availability of exposure or health response relationships for short-term (1–24-h averaged concentrations) exposure and international practice for similar systems. In addition to establishing index values for each pollutant exposure metric over the range of interest, the overall method and algorithm(s) used to calculate the final index should include the effect of the simultaneous exposure to multiple pollutants.

3.2. Application of the index system to monitored data

The context for the alternative API system is the development of the dynamic air pollution prediction system (DAPPS) (Zunckel et al., 2004). The DAPPS was developed by a consortium of four South African partners—CSIR, South African Weather Service, the Peninsula Technikon (now Cape Peninsula University of Technology) and SRK Consulting—and was funded by the Innovation Fund, administered by the National Research Foundation. This project addresses the need for integrated and publicly accessible information on urban scale air pollution, and the communication of the associated potential health impacts through the API system.

4. Results

4.1. Exposure metrics

DAPPS attempts to provide near-to-real-time information on current and short-term future air pollution. The API therefore indicates the likely short-term health impacts of the predicted pollution levels. Thus, pollutants with long-term health effects (health effects that manifest themselves after years to decades of exposure—benzene, 1,3 butadiene, dioxins/furans, polycyclic aromatic compounds, lead, etc.) are excluded from the API calculations. The minimum time resolution of DAPPS is 1 h, so exposures of less than this are also excluded. The pollutants and averaging times included in the DAPPS API are listed in Table 3.

4.2. The definition of the API

The short-term adverse health effects of exposure to the classical air pollutants are essentially respiratory and cardiovascular. The question of the exposure–response relationship for each of the pollutants may be approached from one of the two perspectives: a risk-based approach or a 'toxicological' approach that assumes a threshold below which no adverse effects occur. Of the pollutants under consideration for the API (SO_2, NO_2, PM_{10}, $PM_{2.5}$, O_3 and CO), the PM_{10}, $PM_{2.5}$ and O_3 do not have an apparent threshold value below which the risk of adverse health effect is zero. Continuing research indicates that SO_2 and NO_2 may not have threshold values either (WHO, 2000c, 2005). In other words, except possibly for CO, exposure to these pollutants carries a finite risk of an adverse health effect.

To construct the API, we assumed the availability of appropriate mortality relative risk values RR_i for each of i pollutants. The total attributable risk for simultaneous short-term exposure to several air pollutants is then the sum of the values for each pollutant:

$$(RR-1)_{Total} = \sum_i [(RR_i - 1)], \tag{1}$$

where $i = 1,\ldots,n$ (n is the number of pollutants). In estimating the total risk, care should be taken not to 'double count', for example, not to include 1- *and* 8-h O_3 values.

For convenience, a pollutant sub-index (PSI) is defined to reflect the contribution of individual pollutants to total risk:

$$PSI_j = a_j * (ExposureMetric_j), \tag{2}$$

where the subscript 'j' refers to the pollutant; the ExposureMetric refers to the applicable pollutant-averaging period combination and the coefficients 'a_j' are directly proportional to the incremental risk values (RR_i−1).

We then define an overall API as

$$API = \sum_i PSI_i = \sum_i a_i * C_i, \tag{3}$$

where the C_i are the corresponding time-averaged concentrations.

To illustrate the application of the above-defined API, we used a particular set of RR values (Table 4) for PM, SO_2, O_3 and NO_2 published by the WHO under a procedure for health impact assessment in the EU (WHO, 2001a). These factors, and relative risks for a range of other morbidity and mortality health endpoints, were derived from a meta-analysis of quality-selected time-series studies conducted in

Table 3
Pollutants and averaging periods included in DAPPS API system

Pollutant	SO_2	NO_2	O_3	PM_{10}	$PM_{2.5}$	CO
Averaging periods (h)	1, 24	1, 24	1, 3, 8	24	24	1, 8

Table 4
Relative risk (central estimate) of health outcome per 10 $\mu g\,m^{-3}$ increase in pollutant concentration (WHO, 2001a)

Health endpoint	Incidence per	PM_{10}, 24-h average	$PM_{2.5}$, 24-h average	SO_2, 24-h average	O_3, 8-h maximum	O_3 1-h maximum	NO_2, 1-h maximum
Total Mortality (95% CI)	100000 1013	RR 1.0074 (1.0062–1.0086)	RR 1.015 (1.011–1.019)	RR 1.004 (1.003–1.0048)	RR 1.0051 (1.00023–1.0078)	RR 1.0046 (1.0028–1.0066)	RR 1.003 (1.0018–1.0034)

Table 5
Pollutant sub-indices for DAPPS air pollution index (API) system

Relative risk (RR)	Sub-index value	Concentration corresponding to relative risk value						
		PM_{10}, 24-h average ($\mu g\,m^{-3}$)	$PM_{2.5}$, 24-h average ($\mu g\,m^{-3}$)	SO_2, 24-h average ($\mu g\,m^{-3}$)	O_3, 8-h maximum ($\mu g\,m^{-3}$)	O_3, 1-h maximum ($\mu g\,m^{-3}$)	NO_2, 1-h maximum ($\mu g\,m^{-3}$)	CO, 8-h rolling average ($mg\,m^{-3}$)
1	0	0	0	0	0	0	0	0.0
1.015	1	21	10	38	30	33	51	3.9
1.031	2	41	20	77	60	67	102	7.9
1.046	3	62	30	115	90	100	153	11.8
1.061	4	83	40	153	120	133	204	15.7
1.077	5	104	50	192	150	167	256	19.7
1.092	6	124	60	230	180	200	307	23.6
1.107	7	145	70	268	210	233	358	27.5
1.123	8	166	80	307	241	267	409	31.5
1.138	9	186	90	345	271	300	460	35.4
>1.153	10	>207	>100	>383	>301	>333	>511	>39.3

26 European cities with a combined population of over 30 million. The meta-analysis focussed on single-pollutant model results of all-year studies that included a consideration of short-term ($\leqslant 2$ days) lag effects (WHO, 2004). The health endpoint 'total mortality' is the only one that is common to all the pollutants and time-averaged values under consideration in developing the DAPPS API. CO was not included in the WHO analysis; an RR value of 1.04 (for a 10 ppm increment in exposure), quoted by Schwartz (1995), was used for this pollutant.

The PSIs are essentially numerical values on an arbitrary scale that may be used to facilitate risk communication. To facilitate comparison of our API with the UK system, we assigned a value of '3' to a 1-h maximum ozone concentration of 100 $\mu g\,m^{-3}$ and its corresponding mortality relative risk value. Thus, $PSI_{ozone,1\text{-}h\,max} = 3$ at 100 $\mu g\,m^{-3}$, with corresponding incremental risk $(1-RR_{ozone})$. As incremental risk values for each pollutant are assumed to be constant, this establishes a continuous linear index scale for ozone, with RR = 1 at zero exposure. For each of the other pollutants, the value of the exposure metric (concentration) corresponding to the same RR value RR_{ozone} (1.046, Table 5) is also assigned the PSI value of 3. This then establishes an API that is linear with respect to mortality risk, and internally consistent in the sense that single-pollutant exposure or multiple pollutant exposure with the same total RR would be assigned the same index value.

At a 1-h averaged O_3 concentration of 100 $\mu g\,m^{-3}$ the daily mortality relative risk is 1.046. An O_3 sub-index value of 3 was assigned to this risk level. At this exposure level, the coefficient a_3 (for O_3) is given by Eq. (2): $a_3 = 0.030$. For consistency between pollutant exposure metrics, the exposures (averaging period/concentration combinations) that correspond to the same relative risk (e.g. 1.046) are assigned the same sub-index value, yielding the values given in Table 5. Note that the index values may extend beyond 10 for highly polluted areas.

For simultaneous exposure to the five pollutants under consideration, the API given by Eqs. (2) and (3) is defined in terms of the sum of the mortality risks. For example, at pollution levels C_1–C_5, measured in the appropriate concentration units (Table 6) as $PM_{2.5}$ (24-h average), SO_2 (24-h average), O_3 (1-h maximum), NO_2 (1-h maximum) and CO (8-h maximum), respectively, the corresponding $\text{API} = 0.10C_1 + 0.026C_2 + 0.030C_3 + 0.020C_4 + 0.25C_5$. (The coefficients for the terms C_i are calculated as for the O_3 example above.)

Table 6
Coefficients for calculating pollutant sub-index values

Exposure metric	Coefficient 'a'
PM_{10}, 24-h average	0.048
$PM_{2.5}$, 24-h average	0.10
SO_2, 24-h average	0.026
O_3, 8-h maximum	0.033
O_3, 1-h maximum	0.030
NO_2, 1-h maximum	0.020
CO, 8-h rolling average ($mg\,m^{-3}$)	0.25

Concentrations in $\mu g\,m^{-3}$ unless stated otherwise.

Table 7
Comparison of DAPPS air pollution sub-index values against UK index values

Index value	3		6		9	
Exposure measure ($\mu g\,m^{-3}$)	DAPPS	UK	DAPPS	UK	DAPPS	UK
PM_{10}, 24-h average	62	50	124	75	186	100
$PM_{2.5}$, 24-h average	30	–	60	–	90	–
SO_2, 24-h average	115	–	230	–	345	–
O_3, 8-h maximum	90	100	180	180	271	300
O_3, 1-h maximum	100	100	200	180	300	300
NO_2, 1-h maximum	153	287	307	573	460	700

The close agreement between the DAPPS index values for O_3 and the UK values is by design (Table 7). To harmonise the health-risk messages (descriptors) as closely as possible for at least one pollutant, the DAPPS API value was aligned with the UK 1-h maximum value at an index value of 3, corresponding to the UK Ozone Standard. There are significant differences between the two systems for the other pollutants. We propose a preliminary set of health-risk messages corresponding to API values, and a colour coding system as in Table 8.

The API is intended to convey the total mortality risk associated with simultaneous exposure to the five common pollutants as the sum of corresponding PSI values. It thus provides a ready method of comparing the relative contribution of each pollutant to total risk. This is demonstrated when applied to monitored data in Cape Town for the period 12–14 July 2005 (Fig. 1).

5. Discussion

The practical application of an API system, including the one proposed, in essence attempts to distil and condense a complex body of information into a system capable of communicating as simply and as accurately as possible the health risks associated with a given level of exposure. Inevitably,

Table 8
Proposed breakpoints for health-risk warnings

Total Mortality RR	1– 1.014	1.015– 1.030	1.031– 1.045	1.046– 1.060	1.061– 1.076	1.077– 1.091	1.092– 1.106	1.107– 1.122	1.123– 1.137	1.138– 1.153	> 1.153
API value	0	1	2	3	4	5	6	7	8	9	10
Colour code											
RGB values	0 255 0	154 205 50	255 255 0	255 215 0	255 165 0	255 99 71	255 0 0	139 35 35	205 96 144	139 28 98	139 0 139

LOW (1–3): low risk of increased mortality: 1.5–6.0%.
MODERATE (4–6): moderate risk of increased mortality: 6.1–10.6%.
HIGH (7–9): high risk of increased mortality: 10.7–15.3%.
VERY HIGH (10): very high risk of increased mortality: more than 15.3%.

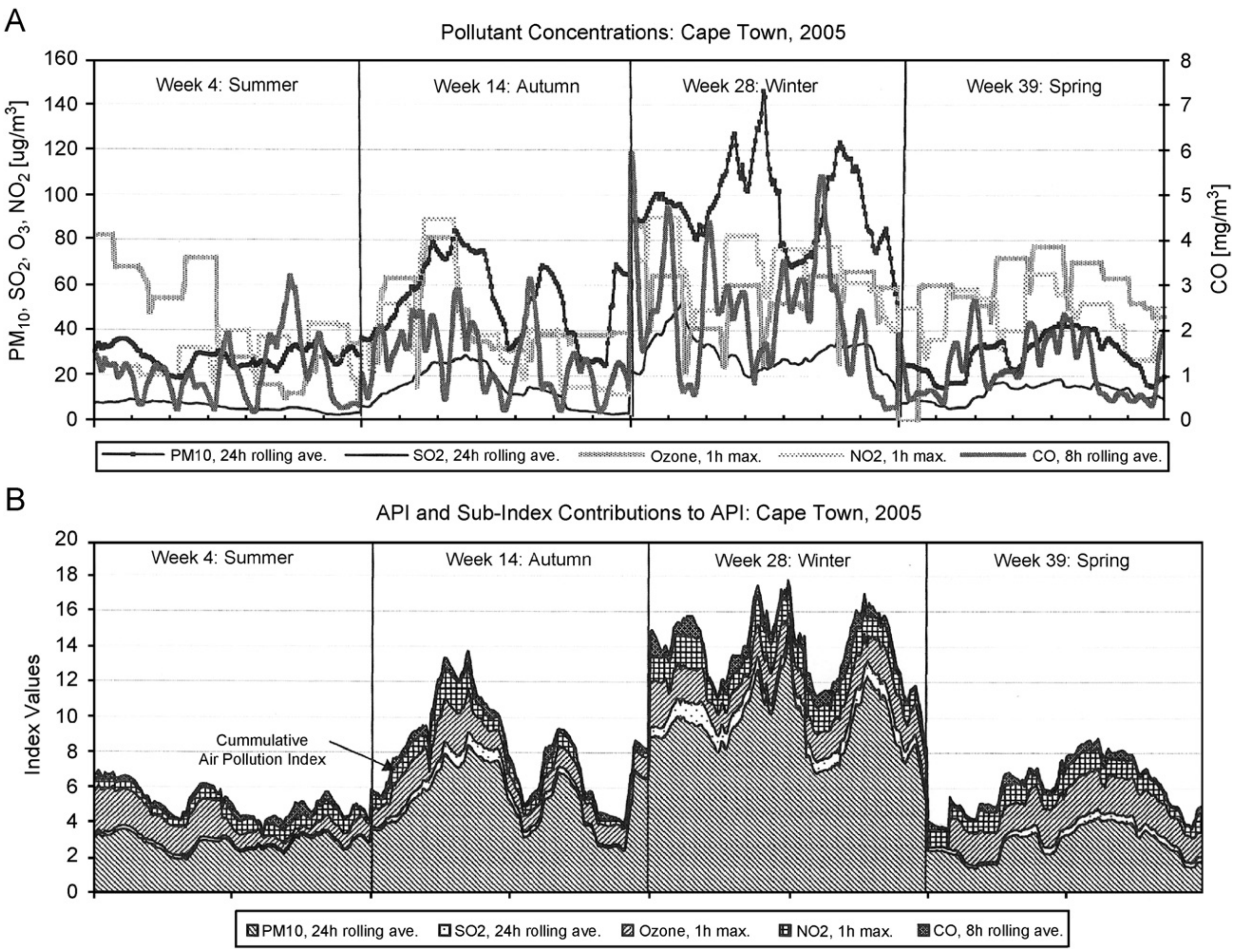

Fig. 1. Application of the DAPPS API to Cape Town monitored data, four 7-day periods representative of seasonal conditions, (A) with the corresponding API values and sub-index contributions to the API (B). (Note that the 'flat spots' in the ozone and NO_2 values occur because these are maximum values in any 24-h period, which change discretely from period to period.)

because of the complexity of the exposure–response relationships, the uncertainties inherent in the quantitative estimate of these relationships on a population basis and the present incomplete state of knowledge, any index system represents a compromise between simplicity and accuracy. The proposed index has the advantage of self-consistency in that a sub-index value for any pollutant included in the index reflects the same increment in relative risk of daily mortality. However, in the absence of a health outcome metric that is capable of combining different morbidity outcomes and/or combined mortality and morbidity outcomes, our API is not necessarily self-consistent with respect to these health outcomes. To address this problem, we investigated applying our approach using disability adjusted life years (DALYs) as the common metric of health effect. In principle DALYs, widely used in burden of disease estimates (Ezzati et al., 2002; Cohen et al., 2005), reduce both mortality and morbidity outcomes to a common factor (the DALY). However, this calculation requires considerable additional information on the health status of the exposed population, and subjective assumptions of age related weighting factors (Fox-Rushby and Hansen, 2001). The use of DALYs to measure the combined effects of mortality and morbidity outcomes is not universally accepted (Anand and Hanson, 1997; Arnesen and Kapiriri, 2004). Nonetheless the development of an API system based on a methodology that uses a health measure of the combined effects of mortality and morbidity outcomes is worthy of further investigation.

The estimation of air pollution health effects, including single-pollutant RR values, given the reality of multi-pollutant exposure, continues to be the subject of ongoing research and debate (examples are Samoli et al., 2005; Dominici, 2002; Schwartz, 2004; McClellan, 2002). The addition of risks derived from single-pollutant statistical models may over-estimate the total effect if pollutant levels are correlated; while models such as the generalised additive model(s) may produce unstable (WHO, 2001b) or heterogeneous estimates (Samoli et al., 2005). Relative risk values of mortality (and other health endpoints) may be different in different regions and/or cities, and, indeed, may be different between different areas of a city, due to factors such as differences in the sources of pollution and/or the chemical composition and size distribution of PM between areas (Schwarze et al., 2006; Bell et al., 2007). The use of surrogate measures of exposure (such as ambient concentrations) introduces uncertainty in the exposure estimate. Thus, it may be some time before consensus on a universal set of independent single-pollutant RR values is achieved, assuming that such a set exists. In the interim, we used a particular set of RR values to illustrate the application of API, recognising that these are conservative in that they may over-estimate the combined risk.

Fig. 1B shows that the proposed API is capable of reflecting observed pollution patterns over typical diurnal and weekly (Monday to Sunday) conditions as well as typical Cape Town seasonal variations—good dispersion conditions during summer, poor dispersion during winter and intermediate/variable conditions during spring and autumn.

The index is clearly sensitive to the RR values used in its construction since individual PSIs are directly proportional to the incremental risk values used. For example, in Fig. 1, PM_{10} is the largest contributor to overall risk and therefore to the API value. If the lower bound mortality PM_{10} RR value of 1.0062 (per $10\,\mu g\,m^{-3}$ increase in 24-h exposure) is used instead of the central value of 1.0074, the PM_{10} contribution to the peak API during the winter period will reduce from 12.7 to 10.6, with a corresponding decrease in the API value, from 17.5 to 15.4.

The proposed API may be compared with the current widely used indices used to communicate short-term adverse health impacts of the common air pollutants. It does not attempt to reflect the total health impact of all air pollutant exposures. Thus, it does not incorporate the possible effects of long-term exposure, nor of pollutants with a latency period (lead, benzene) nor of less widely occurring pollutants with short-term health effects, such as H_2S.

When using monitored time-averaged values or rolling average values or maximum 24-h values, the exposure metric (and hence the API) reflects past exposure (periods between 1 and 24 h). Whilst knowledge of past exposure and potential health effects may be of limited value to effect behaviour change, application of the API to a predictive modelling system (such as DAPPS) enables a prediction of future effects.

There is currently no internationally standardised methodology for constructing APIs. The World Health Organisation methodology for a health-risk assessment associated with air pollution potentially provides a basis for the construction of our API. Further work is required to develop methods for accounting for the influence of city-specific and/or area-specific health-risk effects.

In contrast to the current practice of using a single index value, numerically equal to the highest individual pollutant index values, to reflect the overall air pollution health impact, the proposed API attempts to account for the simultaneous exposure to the common air pollutants, a situation that is ubiquitous.

The proposed index, developed during the DAPPS project as a means to communicate the health risks associated with model-predicted pollutant levels, is self-consistent in that a sub-index value for any pollutant included in the index reflects the same increment in relative risk of daily mortality. The index is a measure of the mortality risk associated with simultaneous exposure to the common air pollutants, and provides a ready method of comparing the relative contribution of each pollutant to total risk. The DAPPS API is also linear with respect to risk, and is capable of reflecting the full range of air pollution associated health risks, from minimal risk in pristine areas to API values >10 for highly polluted conditions.

Acknowledgements

The DAPPS project was funded by the Innovation Fund of the Department of Science and Technology (South Africa), administered by the National Research Foundation. We thank the

Scientific Services Department of the City of Cape Town, who supplied the ambient air quality data. The authors acknowledge the value of the comments of anonymous reviewers of the first draft.

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