



Screening procedure for airborne pollutants emitted from a high-tech industrial complex in Taiwan



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HIGHLIGHTS

- The purpose of this study was to develop a two-stage air dispersion modeling procedure to screen out critical pollutants emitted from an industrial complex.
- The first stage was pre-creating a look-up table of dispersion factor with meteorological data. Secondly, an algorithm was developed to interpolate on the look-up table for dispersion factor with the emission data.
- A “risk strength”, defined as the ratio of concentration to the site boundary standard or air quality standard, was estimated for each air toxic for the screening.
- A total of 1654 records of 21 pollutants emitted from 232 stacks for a high-tech complex site monitored in 2007–2009 were acquired to illustrate this screening method.
- A validation check using ISC3 model with the same meteorological and emission data showed an acceptable overestimate of 6.7% in the average concentration of the 10 nearby receptors.

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ABSTRACT

Despite the modernization of computational techniques, atmospheric dispersion modeling remains a complicated task as it involves the use of large amounts of interrelated data with wide variability. The continuously growing list of regulated air pollutants also increases the difficulty of this task. To address these challenges, this study aimed to develop a screening procedure for a long-term exposure scenario by generating a site-specific lookup table of hourly averaged dispersion factors (χ/Q), which could be evaluated by downwind distance, direction, and effective plume height only. To allow for such simplification, the average plume rise was weighted with the frequency distribution of meteorological data so that the prediction of χ/Q could be decoupled from the meteorological data. To illustrate this procedure, 20 receptors around a high-tech complex in Taiwan were selected. Five consecutive years of hourly meteorological data were acquired to generate a lookup table of χ/Q , as well as two regression formulas of plume rise as functions of downwind distance, buoyancy flux, and stack height. To calculate the concentrations for the selected receptors, a six-step Excel algorithm was programmed with four years of emission records and 10 most critical toxics were screened out. A validation check using Industrial Source Complex (ISC3) model with the same meteorological and emission data showed an acceptable overestimate of 6.7% in the average concentration of 10 nearby receptors. The procedure proposed in this study allows practical and focused emission management for a large industrial complex and can therefore be integrated into an air quality decision-making system.

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1. Introduction

Atmospheric dispersion modeling has been extensively studied since the 1960s and is widely accepted today as an indispensable technique by many air quality managers. Although it is difficult to validate, the technique is employed by many governmental

environmental authorities to evaluate whether emissions of air pollutants from existing or planned sources will comply with ambient air quality standards (Turner, 1994). In the case of environmental impact assessment, the health risk posed by hazardous air pollutants is often assessed. Here, air dispersion modeling provides the basis for estimating excess levels for proposed actions (U.S. EPA, 1989; TEPA, 2011; EPA, 2013c). The dispersion technique is also useful in stack design of emission height and for determination of off-gas exit velocity for worst-case scenarios. However, even with modern computerized techniques, air dispersion modeling for a study site is difficult, as it requires simultaneous input of vast amounts of data with wide variability.

In 2012, the Taiwan Environmental Protection Administration (TEPA) launched a new list of emission standards for air pollutants emitted from industrial stationary sources (TEPA, 2013). The new list consists of 486 chemicals, which are primarily hazardous air pollutants (HAPs) that are regulated by the U.S. EPA (2013a). It includes volatile and semivolatile organic chemicals, pesticides, polycyclic aromatic hydrocarbons, inorganic acids, bases, and heavy metals. Including in the items in this list are 510 regulated air pollutants from industrial sources. In addition to the difficulty of dispersion modeling, the ever-increasing number of air pollutants under regulation poses a significant challenge to both government and industries. However, a full list of assessments may be an overwhelming effort because of the substantial number of pollutants, the large variability in environmental conditions, and the numerous combinations of complex source configurations and receptors (Ma et al., 2012). To overcome these difficulties, this study aimed to develop a simplified procedure to screen air pollutants according to their site boundary standards (S_b), which are critical to the study site. The screening procedure was validated by the Industrial Source Complex of Taiwan EPA. It will be included as a new module in a decision support system of air quality management for complex sites in Taiwan and China (Chiang and Tsai, 2014).

2. Method and mathematical derivation

This study was designed to evaluate the relative impact of various airborne stack emissions and to screen out critical pollutants. Some of the basic assumptions in the development of such a screening method are as follows:

- The predicted concentration of different dispersed chemicals could be normalized by their associated references of regulatory standards or by their threshold limits for cross-pollutant comparison;
- each emission is assumed to originate from a point source and to be continuous;
- all emitted pollutants are inert toward other pollutants during transport.

2.1. Normalization for different pollutants

To develop the screening methodology, the dimensionless risk strength (r_s) is defined as follows:

$$r_s \equiv \frac{\chi}{\chi_{ref}} \quad (1)$$

where

- χ = predicted airborne pollutant concentration at the location of concern (mg/m^3);
- χ_{ref} = reference airborne concentration of the pollutant (mg/m^3);

r_s is conceptually equivalent to a toxicity-based index such as the hazard quotient (HQ) (EPA, 2013a). The χ_{ref} value could be any well-established value, e.g., inhalation reference concentration (RfC) (EPA, 2013b), HAP (EPA, 2013a), or regulatory standard of an air pollutant. It is designed as a normalized measure in order to compare the relative hazards among different pollutants. When a regulatory standard is used for the, χ_{ref} , r_s may not be directly related to health risk, but it could be linked to the relative importance of regulatory concerns.

With a predetermined dispersion factor (χ/Q), the excess airborne pollutant concentration (mg/m^3) attributed to emission sources can be calculated through the expression

$$\chi = \left\langle \frac{\chi}{Q} \right\rangle Q \quad (2)$$

where

- $\left\langle \frac{\chi}{Q} \right\rangle$ = dispersion factor (s/m^3) = concentration per unit of emission rate;
- Q = emission rate of airborne pollutant (mg/s).

With Eq. (1), the r_s value of a pollutant i at location k can be expressed as follows:

$$r_s^{(i,k)} = \sum_j \frac{\chi^{(i,j,k)}}{\chi_{ref}^{(i)}} = \sum_j \frac{\left(\frac{\chi}{Q}\right)^{(j,k)} Q^{i,j}}{\chi_{ref}^{(i)}} \quad (3)$$

where

- $\chi^{(i,j,k)}$ = concentration of pollutant i released from stack j and received at location k (mg/m^3);
- $\left(\frac{\chi}{Q}\right)^{(j,k)}$ = dispersion factor from stack j to location k (s/m^3);
- $Q^{(i,j)}$ = emission rate of pollutant i from stack j (mg/s);
- $\chi_{ref}^{(i)}$ = reference concentration of pollutant i (mg/m^3).

Eq. (3) conceptually implies two steps of computation: (1) calculating the r_s value for pollutant i received at location k from stack j and (2) superimposing r_s values from all stacks. The stack-summed r_s value for pollutant i received at location k can then be ranked according to the r_s value; hence, critical pollutants can be screened out for further study.

2.2. Dispersion factor

For years, air dispersion modeling based on the Gaussian plume theory has been widely used to assess the impact of toxic air emissions on air quality, especially for regulatory compliance. For a ground-level receptor from an elevated release with a defined mixing layer height (z_m), the χ/Q value can be calculated by the classical Gaussian plume formula (Turner, 1994; Napier et al., 2011):

$$\begin{aligned} \frac{\chi}{Q}(x, y, 0) &= \frac{1}{\pi u_e \sigma_y \sigma_z} e^{-\frac{y^2}{2\sigma_y^2}} \left\{ \dots + e^{-\frac{(h_e - 2z_m)^2}{2\sigma_z^2}} + e^{-\frac{(h_e + 0z_m)^2}{2\sigma_z^2}} + e^{-\frac{(h_e + 2z_m)^2}{2\sigma_z^2}} + \dots \right\} \\ &= \frac{1}{\pi u_e \sigma_y \sigma_z} e^{-\frac{y^2}{2\sigma_y^2}} \sum_{j=-\infty}^{\infty} e^{-\frac{(h_e + 2jz_m)^2}{2\sigma_z^2}} \end{aligned} \quad (4)$$

where

- x = downwind distance (m);
- y = crosswind position (m);
- u_e = wind speed at effective release height (m/s);
- σ_y = horizontal dispersion coefficient (m);
- σ_z = vertical dispersion coefficient (m);
- h_e = effective release height (m) = stack height (h_s) + plume rise (Δh) + elevation difference (Δz);

z_m = mixing-layer height (m).

In the above equation, u_e and z_m can often be obtained from meteorological measurements, whereas σ_y and σ_z can be calculated from the z_m value and stability class (SC) from meteorological data.

For most applications, especially in regulatory compliance, χ/Q is calculated from hourly meteorological data. In the case of the long-term average with chronic exposure scenario, the hourly χ/Q values is averaged by the total number of hours (n) (TEPA, 2011):

$$\overline{\left\langle \frac{\chi}{Q} \right\rangle} = \frac{1}{n} \sum_{i=1}^n \left\langle \frac{\chi}{Q} \right\rangle_i \quad (5)$$

2.3. Plume rise

In calculating h_e for Eq. (4), the h_s and Δz values can be obtained directly from the stack geometry and z coordinates. In the case of a negative h_e value (e.g., a receptor at an elevation higher than that for release), ground release ($h_e = 0$) is conservatively assumed. However, the behavior of the Δh value is a rather complex function of emission parameters associated with buoyancy flux (F_b), exit velocity, environmental conditions (e.g., wind speed at stack height, SC), and downwind distance. In the present study, the widely used Briggs (1975) formulas were adopted to calculate Δh (Turner, 1994; EPA, 1995).

In order to decouple the estimated Δh value from meteorological data, a five-year frequency table of wind speed (u_i) and stability class (SC_j) was first compiled for the study site. The categorical table was linearly interpolated among two velocity classes to obtain a continuous frequency distribution function $f(u_i, SC_j)$ for each SC. The average plume rise ($\overline{\Delta h}$) was then calculated by weighting Δh_{ij} for each wind speed category i and SC category j with the frequency distribution:

$$\overline{\Delta h} = \sum_{j=1} \sum_{i=1} \Delta h_{ij} f(u_i, SC_j) \quad (6)$$

where

Δh_{ij} = plume rise at u_i and SC_j (m), calculated from the Briggs formulas;

$f(u_i, SC_j)$ = frequency distribution of wind speed category i and SC category j .

2.4. Screening procedure

The screening procedure for chronic exposure scenarios based on the governing equation for the χ/Q value derived in Section 2.2 is illustrated in Fig. 1. The entire procedure consists of two parts. In the first part, 5-year hourly meteorological data for a study site was acquired. By using the applicable range of h_e values (0–300 m), hourly χ/Q values were calculated through Eqs. (4) and (5) for all directions of winds (16 sectors) at several downwind distances (100–10,000 m). The results were compiled to form a 5-year averaged lookup table $\chi/Q(x, y, h_e)$. At the same time, frequency-weighted plume rises were calculated through Eq. (6) by using several applicable ranges of x , F_b , and h_s values, and the results were regressed to linear equations of $\overline{\Delta h}(x, F_b, h_s)$.

In the second part, four years of emission inventory representative of the study site were acquired. Each selected emission–receptor pair of x and y coordinates and h_e values ($h_e = h_s + \overline{\Delta h} + \Delta z$) were prepared, and all six steps of the procedure below were followed, as illustrated in Fig. 1:

- (1) Interpolate χ/Q for each selected pair from the χ/Q lookup table. To accommodate the large range of χ/Q values over the distance, use a log interpolation of χ/Q .
- (2) Obtain χ by multiplying each associated Q value from the emission inventory over the study period with the interpolated χ/Q value.
- (3) Average χ over the study period.
- (4) Superpose χ values for all emissions at the same receptor.
- (5) Calculate the mean and 95th percentile (P95) χ values for all of the selected receptors.
- (6) For each air pollutant, calculate the mean and P95 of the r_s value through Eq. (1).

2.5. The industrial complex site under study

To demonstrate the screening procedure, the Taichung site of the Central Taiwan Science Park, one of the 13 high-tech parks in Taiwan, was chosen. This site is a world-leading high-tech industrial complex for semiconductors, electronics, and electrical peripherals, with a total investment of over USD 25 billion (CSPA, 2008; IDB, 2014). Besides the conventional pollutants (NO_2 , SO_2 , PM_{10} , O_3), the hazardous pollutants emitted from the site are inorganic acidic (HF, HCl, NH_3 , H_2SO_4) and basic (NH_4OH) chemicals associated with cleaning and etching processes. Fig. 2 shows a site map with a total area of 413 hectares. To the southeast of the site is the Greater Taichung Metropolitan area with a population of 1.5 million. Taichung Industrial Park, a conventional machinery industry complex that has been operating for over three decades, is 5 km south of the site. The Taichung Refuse Incinerator is 3 km southwest, Highway 1 is 2 km east, and Taichung Coal-Burned Power Plant is 15 km east of the site. As the site it is surrounded by these major sources of air emissions, the low-speed southern wind over it is characterized by high levels of background acidic and basic air pollutants in the summer (Chen et al., 2010). The topography of the site is characterized by a low-altitude terrain 50–300 m above sea level, declining from northwest to southeast. The map in Fig. 2 shows 20 representative locations selected for the study. Ten inner receptor locations (D1 to D10; e.g., primary schools, nursing homes, residential buildings) of concern to the public health were selected as regular monitoring stations. Ten outer receptor locations (TC1 to TC10) covering a range of 10 km are required for health risk assessment by the TEPA (2011); for these locations, more stations were selected in the eastern and northeastern part of the Taichung downtown area.

3. Results and discussion

3.1. Meteorological data

Five consecutive years (2006–2010) of hourly meteorological data were obtained from the Taichung Weather Station, as recommended for chronic-exposure scenarios by the Risk Assessment Technical Guideline of 2011 of the TEPA. Fig. 3 shows 16 directions of windrose for 5 years. The north wind in winter was primarily prevailing with a total frequency of slightly less than 15%, where approximately 5% and 8% frequencies were associated with two low-wind-speed categories (1.0–2.0 m/s and 2.0–4.0 m/s, respectively). The frequency distribution for six SCs and five wind speed categories show that a SC (5) and a low wind speed (2 m/s) in the least-dispersion condition had the highest frequency (29%).

3.2. Inventory check and hazard identification

From the inventory check using the Taichung Stationary Emission Database (TEPA, 2012a), 1654 emission records for 21

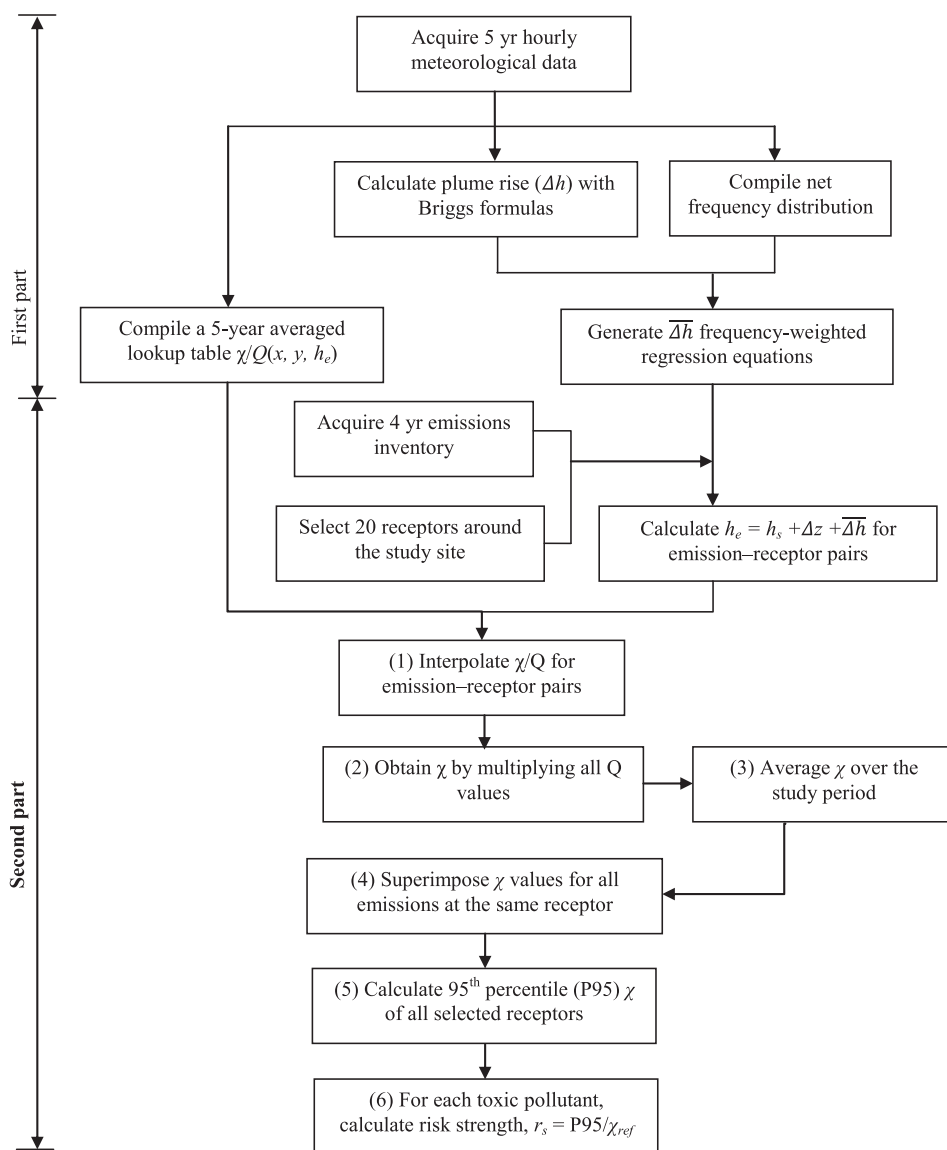


Fig. 1. The screening procedure proposed in this study. χ = airborne concentration, Q = emission rate, χ_{ref} = site boundary standard, h_s = stack height, Δz = elevation difference, $\Delta \bar{h}$ = frequency-weighted plume rise, h_e = effective plume height.

pollutants from 232 stacks from the Taichung site from 2007 to 2009 were identified. From the above inventory check, two tables were compiled, namely, the stack parameter table and emission inventory table. The 21 pollutants include volatile organic chemicals (VOCs), PM_{10} , and dust, which are aggregate pollutants. These were eliminated from further analysis because of lack of reference data. Table 1 lists the remaining 18 pollutants for screening analysis. Among these, carbon monoxide (up to 192 t/yr), nitrogen oxide (up to 27.06 t/yr), and methylbenzene (up to 40.2 t/yr) were emitted at much higher rates. There were nine identified acids and bases. Sulfuric acid was emitted at the highest rate (16.31 t/yr). The four criteria pollutants CO, SO_2 , NO_2 , and Pb, were also identified.

The two types of reference concentrations that were identified for this study are reference concentration (RFC) and site boundary standard (S_b). Traditionally, many applications assess HQ relative to RFC to determine non-carcinogenic risk in chronic exposure scenarios. However, only nine RFC values could be obtained from the EPA (2013b) Integrated Risk Information System and DOE (2013) Risk Assessment Information System databases. Therefore, S_b values from the Emission Standards of Stationary Pollution Sources (TEPA, 2013) were used as references to evaluate r_s values. For

the four criteria pollutants, ambient air quality standards (TEPA, 2012b) were used as references instead. Table 1 shows that arsine and phosphine are the two most hazardous chemicals, with S_b values on the order of 10^{-3} mg/m³, which are lower than those for chlorine gas, hydrofluoric acid, lead, phosphoric acid, sodium hydroxide, sulfuric acid and sulfuric acid droplet (on the order of 10^{-2} mg/m³).

3.3. Plume rise regression formulas

A plume generally rises to equilibrium after a few hundred meters of downwind distance (x). Accordingly, the frequency-weighted heights of plume rises were linearly regressed into two best-fit formulas using x , h_s , and F_b values:

$$\Delta \bar{h} = \begin{cases} 31.1 + 0.192x - 0.661h_s + 1.40F_b & \text{for } x \leq 300 \text{ m} \\ 66.5 + 3.45 \times 10^{-5}x - 0.734h_s + 1.99F_b & \text{for } x > 300 \text{ m} \end{cases} \quad (7)$$

The R^2 statistics values show fair linearity for both equations, with slightly higher linearity (0.8678 vs. 0.9536) for the long distance with more observations (495 vs. 1320). The average plume

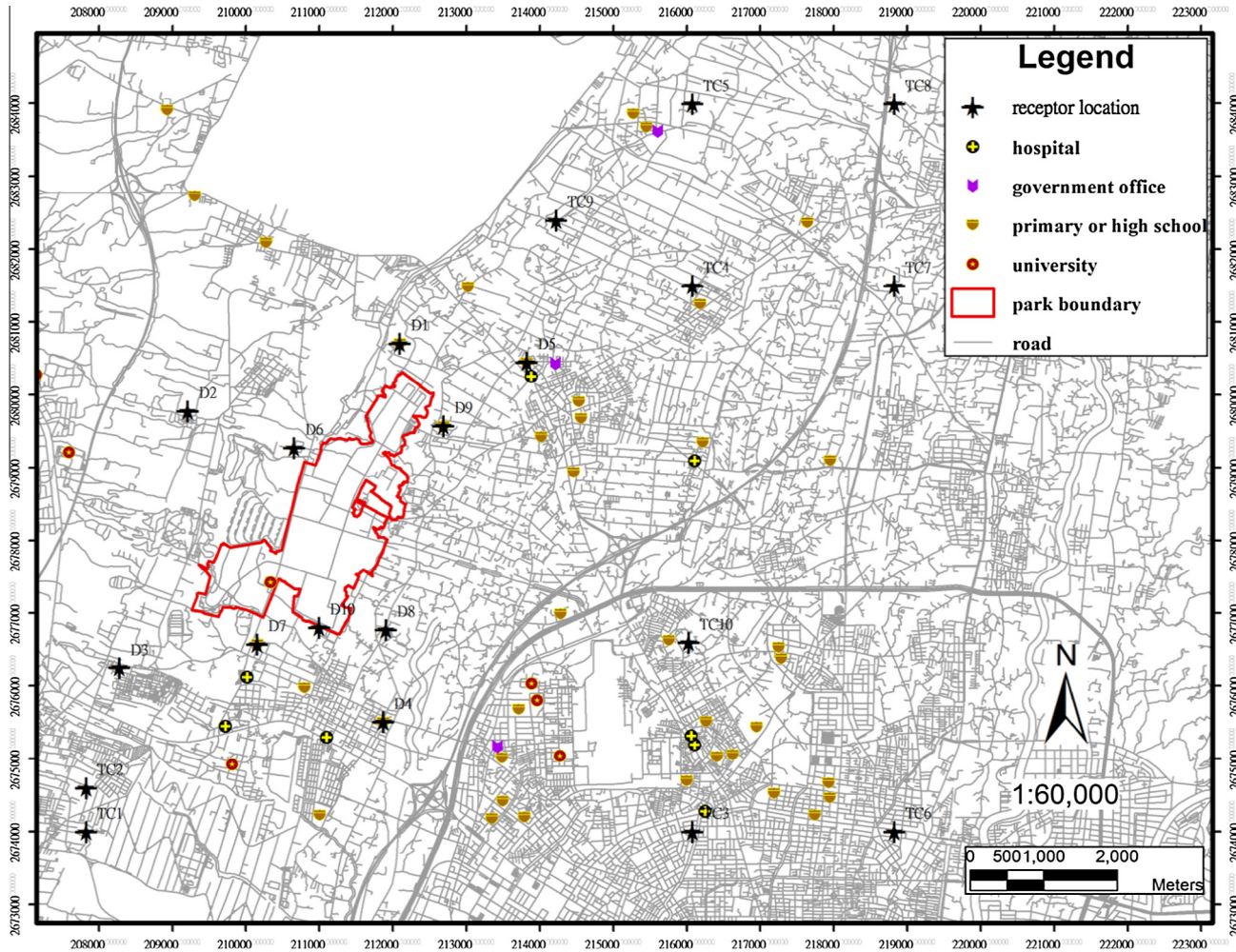


Fig. 2. Map of the high-tech industrial complex showing 10 inner receptor locations (D1–D10) of regular monitoring stations and 10 outer receptor locations (TC1–TC10) covering a ~10 km range required for health risk assessment by the TEPA.

height is linearly proportional to x and F_b values but inversely proportional to h_s . Within the 300 m proximity, a transitional plume rise may occur, and the regression yields a higher relative error for the intercept (7% vs. 2%). Compared with the values of intercepts, the x coefficients are much smaller; this suggests an insignificant contribution to the prediction of plume heights for the long range. In addition, the high p value supports the high likelihood of zero coefficients for x ; accordingly, x could be reasonably eliminated from the regression formula.

3.4. Calculated χ/Q value

Dispersion coefficients and emission rates are the two major variables that affect the modeling results of a general steady-state doubled Gaussian plume mathematical equation on which most air dispersion plume modeling is based. For chronic-exposure scenarios, dispersion coefficients are associated with long-term meteorological data and emission data with a representative operation in a given period. As illustrated in the screening algorithm in Fig. 1, the first part is the most computationally intensive because of the significant data-crunching processes. With the precalculated χ/Q lookup table and the regression formula for plume rise, the second part of χ/Q estimation at selected emission–receptor pairs can be significantly simplified. For the screening procedure developed in this study, the following mechanisms in the Gaussian plume formula were not considered: dry

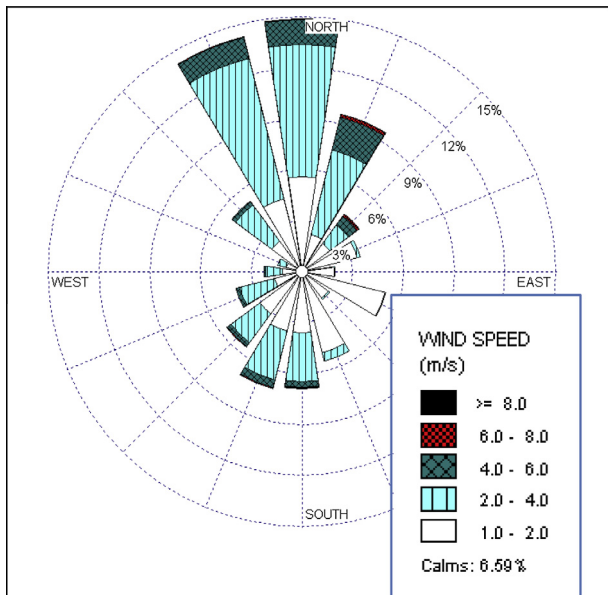


Fig. 3. Sixteen directions and five wind speeds of windrose for the 2006–2010 hourly data for the study site.

Table 1

Results of the inventory check for airborne pollutants in the Taichung site from the emission database for 2007–2010.

No	Name	Chemical formula	Emission rate (t/yr)	RfC ^a (mg/m ³)	Site boundary standard ^b (mg/m ³)
1	Acetic acid	CH ₃ COOH	0.007–0.4	N/A	5.00E–01
2	Ammonia	NH ₃	0.001–4.66	1.00E–01	6.95E–01
3	Arsine	AsH ₃	0.001–0.12	5.00E–05	4.00E–03
4	Carbon monoxide	CO	0.002–192	N/A	4.01E+01 ^c
5	Chlorine gas	Cl ₂	0.001–2.409	N/A	5.81E–02
6	Dimethylbenzene (xylene)	C ₈ H ₁₀	0.246–0.246	N/A	8.67E+00
7	Hydrofluoric acid	HF	0.001–0.713	1.40E–02	1.00E–02
8	Hydrogen chloride	HCl	0.001–1.174	2.00E–02	1.49E–01
9	Lead and compounds	Pb and PbX	0.001–0.001	N/A	1.00E–02 ^c
10	Methylbenzene (toluene)	C ₇ H ₈	1.86–40.2	5	7.53E+00
11	nitric acid	HNO ₃	0.001–0.604	N/A	1.00E–01
12	Nitrogen oxides	NO ₂	0.001–27.06	N/A	4.50E–01 ^c
13	Phosphine	PH ₃	0.001–0.12	3.00E–04	8.00E–03
14	Phosphoric acid	H ₃ PO ₄	0.001–0.071	1.00E–02	2.00E–02
15	Sodium hydroxide	NaOH	0.232–0.232	N/A	4.00E–02
16	Sulfur oxides	SO ₂	0.002–16.31	N/A	7.85E–01 ^c
17	Sulfuric acid	H ₂ SO _{4(g)}	0.001–16.3	1.00E–03	2.00E–02
18	Sulfuric acid droplet	H ₂ SO _{4(l)}	0.001	1.00E–03	5.00E–02

^a Integrated Risk Information System (EPA, 2013b) and Risk Assessment Information System (DOE, 2013).^b Emission Standards of Stationary Sources (TEPA, 2013).^c Ambient Air Quality Standards (TEPA, 2012b).

deposition, wet deposition, and chemical reaction. Consequently, the predicted values of χ/Q and r_s were conservatively estimated.

Two important intermediate results before the final calculation of r_s values are the χ/Q lookup table and frequency-weighted plume rise regression formula. A series of 3-D wireframe plots of log scale of χ/Q were presented for h_e values of 0, 60, 200 m (Fig. 4) over the range of x and y Cartesian coordinates for up to ± 10 km. To examine the detailed changes, the distance scale was magnified to ± 600 m, where the changes were more significant. A significantly steep change around the release origin within a few hundred meters even on the log scale of χ/Q could be observed. Beyond the first kilometer, the changes for all release heights were relatively small.

Upon comparison of the χ/Q values with different h_e values on the ± 10 km scale (Fig. 4), it is apparent that lower h_e values are associated with higher χ/Q plateaus (ranging from 10^{-8} to 10^{-5} s/m³), in accordance with the Gaussian dispersion theory. The magnified scale of ± 600 m predicts the χ/Q results of the two higher h_e values and shows the depressed patterns around the origin. The result corresponding to $h_e = 0$ m shows an elevated pattern. It should be noted that the predicted χ/Q results for emission characteristics and meteorological conditions are site-specific. Furthermore, when using the χ/Q table, only h_e and x - y coordinates of interest are required to estimate χ/Q values from the contour plots of the study site in Fig. 4.

In this study, all calculations with dispersion equations were programmed in Microsoft Excel. Alternatively, the χ/Q lookup table and regression formulas could be generated through other acceptable methods such as ISCST3, GENII, AERMOD, or any regulation-compliant dispersion codes. Although the aforementioned codes may utilize more sophisticated built-in mechanisms of dispersion, the procedure still requires little or no modification because its lookup table and regression are essentially just an interpolation algorithm. To validate the procedure and to check the results, the same set of meteorological data and emission inventory was used in TEPA's recommended ISCST3. The average concentration at the 20 receptor locations was 19% less than that of the screening procedure (results from the procedure are conservative). With only the inner locations (D1 to D10) considered, the ISCST3's average is only 6.7% lower. Thus, results from the full-strength industrial dispersion code show reasonable accuracy and reliable precision regardless of the simplifying approach used

in this study. In addition, the Excel implementation in this study is flexible enough to accommodate various data sets and equations, and is transparent and traceable in step-by-step calculations.

3.5. Ranking results

Fig. 5 presents the ranking results for 10 air pollutants in this study. The P95 and mean r_s values for the 20 selected locations fall in the range of 10^{-2} – 10^{-4} and 10^{-2} – 10^{-5} , respectively. The P95 value is used as an upper bound to protect the general public residing around the site in a chronic scenario (TEPA, 2011). The top four pollutants in terms of P95 value were nitrogen dioxide, hydrogen fluoride, chlorine, and sulfuric acid droplet, the last three of which are directly associated with industrial activities. These three pollutants could serve as indicators for air toxics for intensive monitoring and further analysis. D6, on the northwest boundary of the Taichung site, had the highest r_s value for nitrogen dioxide; it is about 100 m away from the nearest release point. As shown in Fig. 3, this hot spot is expected because of the prevailing southeast wind.

Although Table 1 does not include VOC, an aggregate pollutant, 15 of the identified individual VOC showed a significant increase in ambient concentrations around the Taichung site (CSPA, 2008). However, the emission sources in this area are complicated (see Section 2.5). Xylene and toluene are among the 15 VOCs listed in Table 1. If all the VOC were summed together for screening, then the P95 r_s value of the VOCs would be ranked in the top 5.

The lookup table for χ/Q and the regression formulas for Δh were created by using 5 years of hourly meteorological data to cope with the two most computationally intensive steps using Eqs. (4)–(7). The table and formulas were developed beforehand to decouple the release–receptor data from the meteorological data. The table and regression formulas could be used as long-term steady dispersion systems for site-specific environmental settings. Even with different sets of emission data in different years of release inventory for retrospective or perspective prediction, different receptor locations, or different regulated pollutants, the same screening procedure can be applied with little or no modification. Although the demonstration in this study is based on simple point sources, the χ/Q interpolation could be readily extended to complicated large-area sources (Yuan et al., 1988).

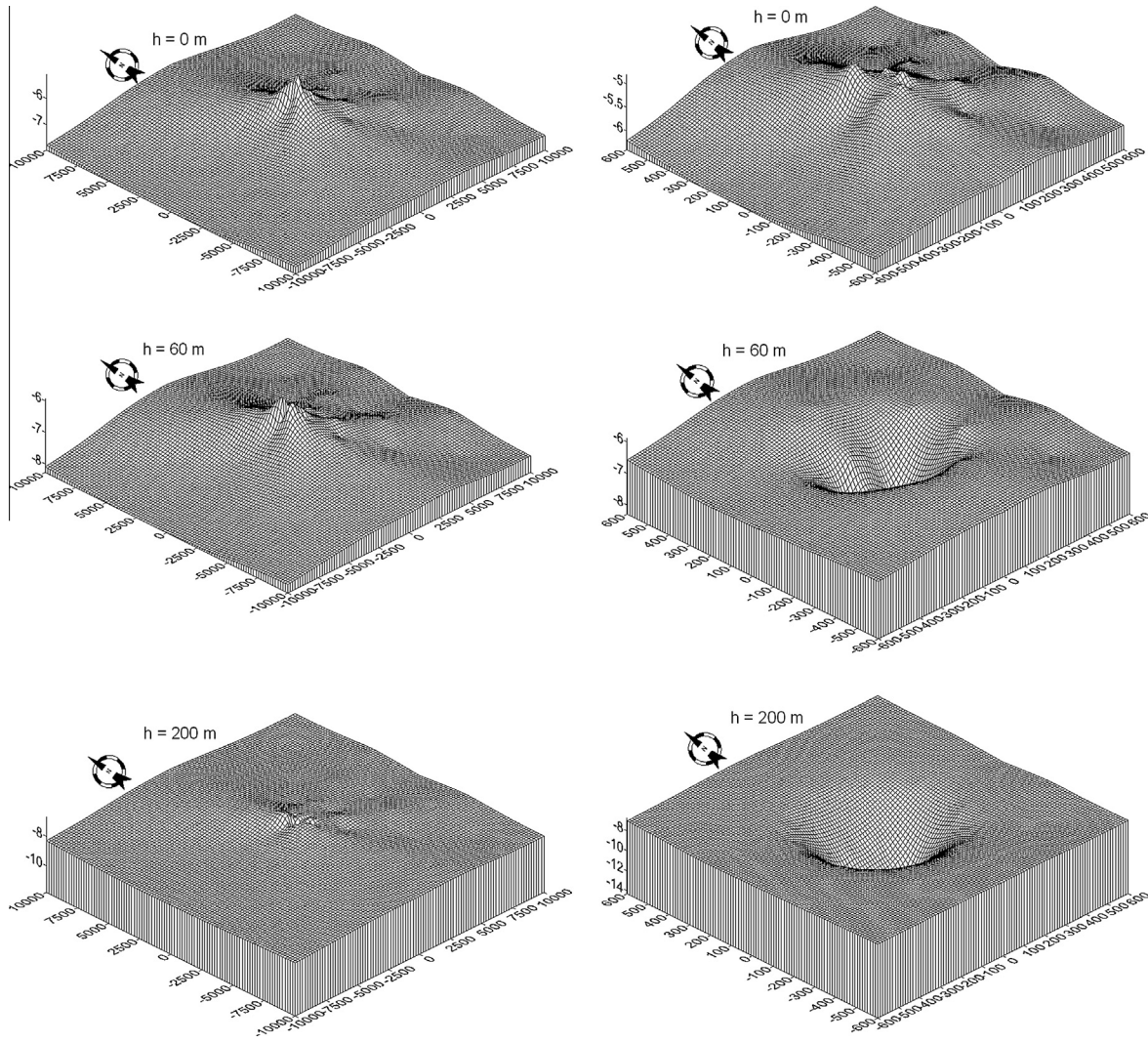
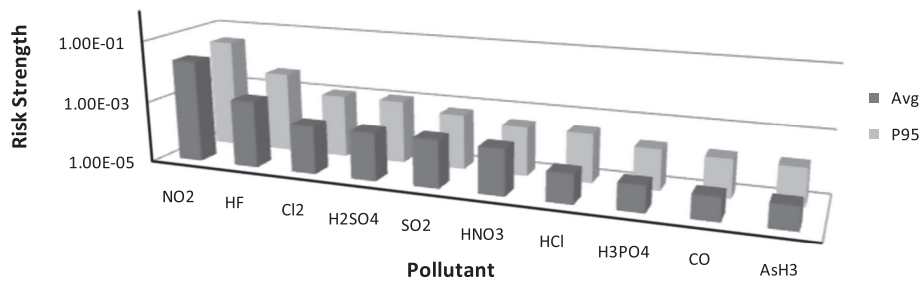


Fig. 4. Results of 3-D wireframe plots of predicted logarithmic average of χ/Q for the Taichung site for the period 2006–2010. Analysis utilized hourly meteorological data for three effective release heights (0, 60, 200 m). The left plot covers 10 km, while the right plot covers closer distances (up to 600 m) to show detailed changes in the $\log z$ scale of χ/Q around the source in the center.



	NO2	HF	Cl2	H2SO4	SO2	HNO3	HCl	H3PO4	CO	AsH3
Avg	2.10E-02	1.51E-03	3.61E-04	3.52E-04	3.42E-04	2.96E-04	8.67E-05	6.83E-05	5.47E-05	5.45E-05
P95	3.56E-02	4.15E-03	1.09E-03	1.09E-03	6.01E-04	3.82E-04	3.74E-04	1.98E-04	1.66E-04	1.53E-04

Fig. 5. Screening results for the mean and 95th percentile r_s values for the ten highest values for the 20 selected receptor locations of the Taichung site.

4. Conclusions

To meet regulatory requirements for the continuously increasing amount of airborne pollutants in high-tech science parks, a

practical and sufficiently accurate screening procedure was developed in this study. The operationally defined r_s value, as well as the design of the χ/Q lookup table and regression formulas, allows the screening of critical pollutants with various long-term exposure

scenarios and applications. This procedure is particularly advantageous for in-house decision making for decoupling of extensive Gaussian plume calculations with meteorological data from large arrays of emission data.

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