

# Chapter 2

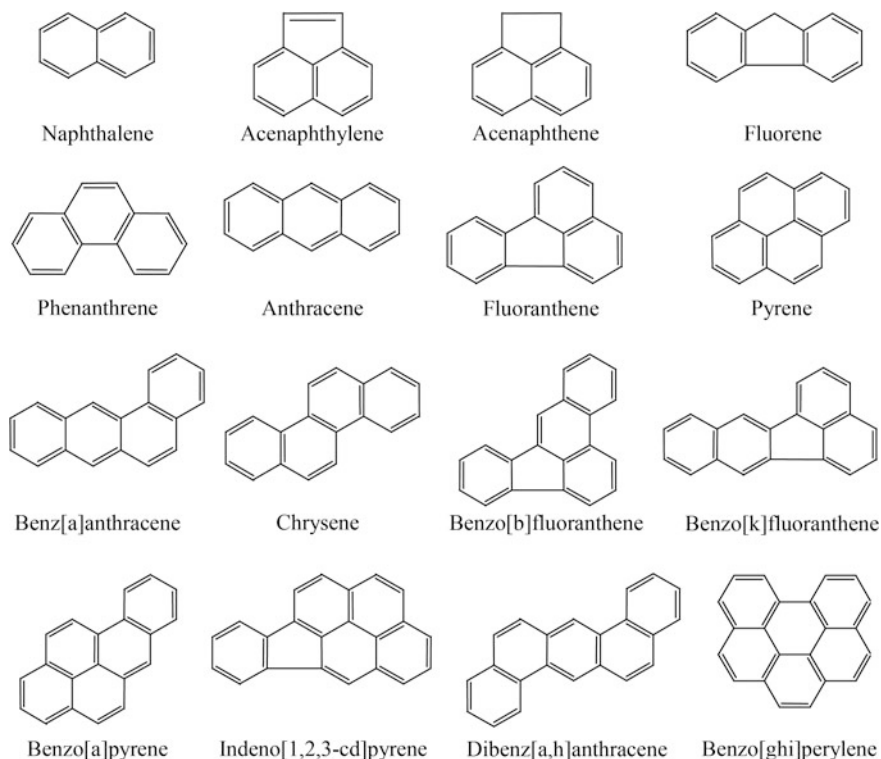
## Research Background

### 2.1 Introduction of Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are a class of organic chemicals, which include carbon and hydrogen with a fused ring structure containing at least 2 benzene rings (Baek et al. 1991; Ravindra et al. 2008). Due to their persistency and long-range transport capacity, PAHs are ubiquitous in ambient air (Lunde and Bjørseth 1977; Wang et al. 2010). PAHs are of great concerns as these compounds were one of the first atmospheric pollutants that were identified as being carcinogenic (Boffetta et al. 1997; Armstrong et al. 2004). Recently, benzo[*a*]pyrene (BaP), one of the high molecular weight PAHs, has been classified into the group of most carcinogenic agents by the International Agency for Research on Cancer (IACR 2014). Atmospheric PAHs are mostly emitted by incomplete combustion of carbonaceous materials, including both anthropogenic activities and natural sources, such as indoor solid fuel burning, waste burning, vehicle exhausts, coke production, industrial boilers, and deforestation and wildfires (Ravindra et al. 2008).

Because of their low vapor pressure, some PAHs are present in both gas and particle phases (adsorbed on particles) in air. The lower molecular weight PAHs, such as naphthalene, acenaphthene, and phenanthrene, are found almost in gas phase, while the higher molecular weight PAHs, such as benzo[*a*]pyrene, are almost exclusively in particle phase. The molecular structures and physiochemical properties of sixteen parent-PAH compounds are listed in Fig. 2.1 and Table 2.1.

In the environment, PAHs are usually present as mixtures with relatively higher melting and boiling points. Because of their distinguished structures, different PAH compounds have different physiochemical properties. Compounds with similar molecular weights and rings exhibit similar physiochemical properties and environmental behaviors. Generally, as rings and molecular weights increase, their solubility, saturation vapor pressure, and Henry's constants decrease leading to the decrease of volatility and water solubility, whereas octanol–water partition coefficient and lipophilicity increase (Smith et al. 1999).



**Fig. 2.1** The molecular structures of sixteen priority-listed PAHs

## 2.2 Emissions of Polycyclic Aromatic Hydrocarbons

Major PAH emission sources have been identified thanks to numerous studies on PAH emission characteristics. Incomplete combustion of carbonaceous materials such as biomass burning in residential stoves can lead to the release of high levels of PAHs. Other processes such as pyrolytic conditions in coking plants and oxidation reduction reactions in primary aluminum production are also conducive to PAH formation (Ravindra et al. 2008). Although natural sources contribute a part of atmospheric PAH emissions, anthropogenic activities contribute the majority. Not only residential and industrial sources but also motor vehicles, ships, and agricultural sources are responsible for PAH contamination. Although there were many factor affecting emission factor values of different sources, most of them can be classified into two categories: combustion efficiency and emission control device. Lower combustion efficiency leads to much more rapid PAH formation during combustion, and emission control devices with lower PM remove efficiency also release more particle-phase PAHs.

**Table 2.1** The physiochemical properties of sixteen priority-listed PAHs

Compound	SN	Rings	MW	BP	MP	S	$V_p$	H	$\lg K_{ow}$
Naphthalene	NAP	2	128.18	209	80	31.5	$1.1 \times 10^1$	43.0	3.37
Acenaphthylene	ACY	2	152.20	290	124	3.93	$8.9 \times 10^{-1}$	11.55	4.00
Acenaphthene	ACE	2	154.20	252	108	3.93	$2.9 \times 10^{-1}$	24.0	3.92
Fluorene	FLO	2	166.23	276	119	1.98	$8.0 \times 10^{-2}$	8.50	4.18
Phenanthrene	PHE	3	178.24	326	136	1.15	$2.5 \times 10^{-2}$	4.0	4.57
Anthracene	ANT	3	178.24	326	136	0.075	$1.1 \times 10^{-3}$	6.0	4.54
Fluoranthene	FLA	3	202.26	369	166	0.206	$1.1 \times 10^{-3}$	0.659	5.22
Pyrene	PYR	4	202.26	369	166	0.132	$5.5 \times 10^{-4}$	1.10	5.18
Benz(a)anthracene	BaA	4	228.30	400	177	0.009	$1.5 \times 10^{-5}$	0.102	5.91
Chrysene	CHR	4	228.30	400	177	0.002	$6.1 \times 10^{-7}$	0.106	5.86
Benzo(b)fluoranthene	BbF	4	252.32	461	209	0.002	$2.1 \times 10^{-5}$	0.054	5.80
Benzo(k)fluoranthene	BkF	4	252.32	430	194	0.0008	$1.3 \times 10^{-7}$	0.111	6.00
Benzo(a)pyrene	BaP	5	252.32	461	209	0.004	$7.5 \times 10^{-7}$	0.009	6.04
Indeno(1,2,3-cd)pyrene	IcdP	5	276.34	498	233	0.0005	$1.0 \times 10^{-10}$	N/A	6.50
Dibenz(a,h)anthracene	DahA	5	278.36	487	218	0.0006	$4.3 \times 10^{-10}$	0.007	6.75
Benzo(g,h,i)perylene	BghiP	6	276.34	467	218	0.0003	$1.4 \times 10^{-8}$	0.001	6.50

*Note* SN short name; MW molecular weight; BP boiling point (°C); MP melting point (°C); S water solubility at 25 °C (mg/L);  $V_p$  saturation vapor pressure at 25 °C (Pa); H Henry's constant (Pa·m<sup>3</sup>/mol)

**Residential sources.** Residential sources including residential burning of firewood, straw, coal, garbage, or other organic substances within fireplace, woodstoves, and other devices for heating, cooking, and lighting contribute the largest part of PAH emissions. Generally, these devices are lack of exhaust control treatment, and stove design, fuel types and characteristics, and operational practice determine the combustion efficiency and consequently PAH emission rates. Modification of air flow control, thermal control and heat storage, and usage of combustion catalysts can lead to reduced PAH formation and release (Mead et al. 1986; Kelly 1983). In terms of fuel types, coal, oil, and gas are all associated with PAH emissions, but burning of gas shows a much lower emission level. Residential biomass burning represents the highest PAH emission rates among all combustion sources and fuel types. In developing countries, solid biomass including wood, straw, and dung cakes is widely used for cooking and heating because they are cheaper and much easier to access. About 3 billion residents are using solid fuels for daily cooking globally, and among them a large fraction use solid biomass. Therefore, residential biomass burning is of great concerns in terms of PAH exposure and human health (WHO 2002).

**Transportation sources.** Transportation sources include exhaust emissions from on-road motor vehicles, ships, and aircrafts, among which on-road motor vehicles are the most important sources in terms of either emission amount or proximity to people. In urban areas, a great deal of PAH contaminations are associated with vehicle emissions which include emissions from both oil burning in internal-combustion engines and leaking of unburned fuels and lubricant (USEPA 1998). Emission factor levels of motor vehicles depend on engine type, load, fuel type and quality, driving mode (such as cold or warm start), and exhaust control device. Emission factors of different vehicle types are quite different. Generally, heavy duty vehicles have higher emission factor values than light duty ones, and emission factors of diesel vehicles tend to be higher than those of gasoline ones (Riddle et al. 2007). Driving mode is also an importing factor influencing PAH emissions. Compared to driving with constant speed, congested traffic conditions with vehicles only traveling short journeys enhance PAH emissions significantly (Kado et al. 2005). Additionally, emission factors measured using different methods could be very different. For example, it was reported that emission factors derived from real-world test were significantly higher than those from dynamometer tests (Kristensson et al. 2004), and emission factors observed in a tunnel test were considerably higher than those measured in a roadside test (Wingfors et al. 2001). Other factors including age, load, and lubricant oil also lead to differences of emission factors of motor vehicles. Although factors affecting vehicle emissions are complicated, it is the implementation of emission standards that actually drives the descending tendency of emission factors over time. For instance, from 1992 to 2014, six emission standards (Euro I to Euro VI) have been carried out among European Union (Timilsina and Dulal 2009), while in the USA, an important step toward vehicle emission control was taken in 1970 when the Congress passed the Clean Air Act, which was further amended in 1977 and 1990 (Timilsina and Dulal 2009). Development of most new emission control technologies was primarily driven by these regulations. The first-generation catalytic converter introduced in the mid-1970s helped to cut car emission substantially (USEPA 1999). After the three-way catalysts with on-board computer and oxygen sensor hit the market in the 1980s, more evident reduction in car emission was achieved (USEPA 1999). Although PAH emission was not regulated directly, the new technologies aiming at other pollutants including particulates and nitrogen oxide helped to trim down PAH emission “unintentionally.” Except for exhaust control devices, emission standards lead to technical improvement of vehicle engines at the same time. The increase of combustion efficiency directly limits PAH formation and further reduces PAH release in the exhaust. For the same reason, vehicle emission was also gradually reduced in developing countries over years, although the progress was hysteretic (ADB 2003). For example, China V emission standard, which is similar to Euro V, already went into effect in China in 2015.

**Industrial sources.** Industrial sources include emissions from power plants and industrial boilers, leaking from coke and primary aluminum production processes, iron and steel industry, and petroleum industry. Given that industrial boilers are often equipped with emission control devices, factors determining PAH emissions

from these boilers involve both boiler combustion efficiency and types of control devices. Control devices with higher PM remove efficiency can remove PAHs more effectively, especially those compounds with high molecular weights (USEPA 1998). Generally, the bag dust removal has the highest remove efficiency following the electrostatic precipitation, wet dust removal, and cyclone dust removal. In developed countries, bag dust removal has been widely applied to the assistance of one or several other devices, while in developing countries electrostatic precipitation is still the most popular control device for industrial boilers. Despite all this, given the much higher combustion efficiency and remove efficiency, industrial boilers contribute only small fractions of total atmospheric PAH emissions both in developed countries and in developing countries (Zhang and Tao 2009). Coking industry can be divided into mechanical coking and beehive coking with regard to PAH emissions. Emissions from coke production are mainly due to leak from coke oven plants in which the pyrolytic process that facilitates PAH formation occurs. Beehive coking which has been seldom seen in developed countries is found to be associated with high levels of PAH emissions because of the poor design of coke ovens and lack of control devices. Beehive coking had spread without restriction in China in the 1990s, especially in Shanxi and Guizhou provinces (Zhang et al. 2007). The coal law promoted in the late 1990s prohibited beehive coking on a national scale (Law of the People's Republic of China 1996). However, since this coking activity is quite difficult to be found out, beehive coking has never been eradicated thoroughly in China. Although better than beehive coking in terms of PAH release, mechanical coking also represents higher emission factors. Since gas in the coke ovens contains high levels of PAH, leaking around the ovens is associated with severe release of PAHs into ambient air. The processes that can induce emission include leaking from charging, pushing, quenching, doors and topside, as well as combustion-related emissions from battery stacks (USEPA 2011). Control devices have been applied to these leaking points, which can significantly reduce emissions. However, leaks cannot be entirely avoided, and as a result, coking industry is responsible for a large fraction of PAH emissions. Regarding the type of reduction cell used, primary aluminum production can be divided into prebaked and Soderberg (USEPA 2011). Prebaked technology came out later than Soderberg. It is expected that PAHs have already been released during anodes prebaking process, when the emissions can be easily controlled; the emission from reduction of prebaked technology are much less than those from Soderberg cells (USEPA 2011). Addressing different technology and control devices associated with different industry activities is crucial for reducing uncertainty of PAH emission estimation.

**Agricultural and natural sources.** Agricultural and natural sources mainly include emissions from agricultural machinery, open burning of agricultural waste, deforestation, savanna, forest, and peat fires (Ravindra et al. 2008; Zhang and Tao 2009). PAHs are emitted from fossil fuel combustion in internal engines of agricultural machinery. The emissions are similar to those of on-road motor vehicles. Agricultural waste open burning and deforestation are common methods for residue disposal and land preparation. These activities involve burning of organic matters under suboptimum combustion conditions and thus lead to a large amount of PAH

emissions (Ravindra et al. 2008). Similar combustion conditions can be found during savanna, forest, and peat fires. Studies reveal that vegetation fires contribute more than 10 % to total PAH exposure in sub-Saharan Africa (Lammel et al. 2013). High contributions from these sources can be also expected in South America and Southeast Asia where vegetation and peat fires are widespread.

Development of PAH emission inventory is essential to source appointment, transport modeling, and exposure and health risk assessment of these compounds. Many studies have been carried out to address PAH emissions on regional to global scales. Emission inventories have been established for countries such as the USA, the United Kingdom, the former Soviet Union, China, and regions such as North America, Europe, Asia, as well as the globe (USEPA 2011; NAEI 2011; EMEP 2011; van der Gon et al. 2007; Galarneau et al. 2007; Xu et al. 2006; Zhang et al. 2009). Estimates from some of these studies are listed in Table 2.2. Emissions from transportation sources often contribute a significant fraction in developed countries, while in developing countries biomass in residential sector is commonly dominant. In a previous study, global emissions of 16 PAH compounds were estimated to be 520 Gg in 2004. The contributions of residential biomass burning and wildfires are 56.7 and 17.0 %, respectively. China (114 Gg), India (90 Gg), and the USA (32 Gg) are three countries with the highest total emissions (Zhang et al. 2009). This is the only study that addressed PAH emissions on a global scale. However, being limited by activity data and emission factors, only 1-year country-level emissions were reported without spatial information. Hence, establishment of spatially resolved and temporally informed emission inventory is expected for transport modeling and health risk assessment.

Methods to estimate emissions of other compounds are instructional for PAH emission estimation. For instance, in a previous study, a technology split method

**Table 2.2** Comparison of PAH emission estimations from various emission inventories

Region	Period	Compound	Emission, Gg/y	Reference
Former Soviet Union	1990–1997	6 PAHs	1.02	Tsibulsky et al. (2001)
Europe	1970–1995	BaP	0.59	Pacyna et al. (2003)
Europe	1990	6 PAHs	12.5	Berdowski et al. (1997)
Europe	1990	4 PAHs	2.4	EMEP (2011)
U.K.	1970–2008	16 PAHs	1.56	NAEI (2011)
U.K.	1995	16 PAHs	3.8	Wenborn et al. (1999)
Great Lakes	2002	BaP	0.0268	Great Lakes Comm. (2007)
U.S.A	1990	16 PAHs	26.5	USEPA (2011)
North America	2002	6 PAHs	18.2	Galarneau et al. (2007)
China	1980–2003	16 PAHs	25.3	Xu et al. (2006)
Globe	1966–1969	BaP	5	Suess (1976)
Globe	2004	16 PAHs	520 Gg	Zhang et al. (2009)

has been applied to estimate global emission of black carbon (Bond et al. 2004). This method employs Gauss curve to simulate temporal trends of technology diffusion such as the application of new engine designs, new industrial processes, or highly efficient control devices. This method is also suitable for PAHs and should significantly reduce uncertainty in emission estimation.

## 2.3 Global Transport Modeling of Polycyclic Aromatic Hydrocarbons

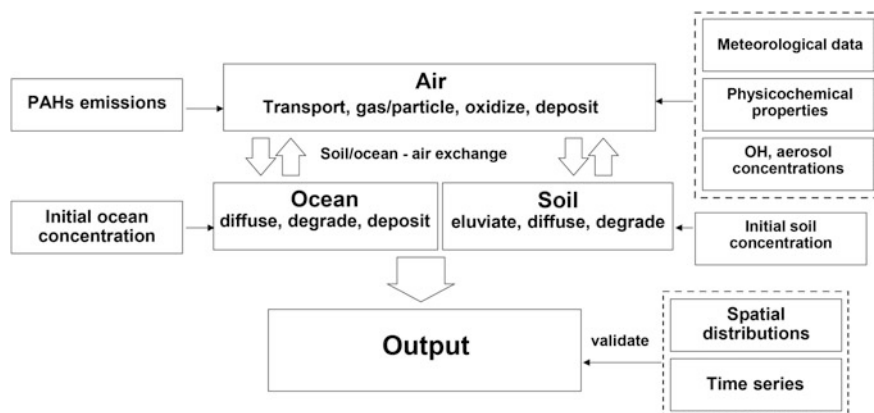
As early as 1960s, the long-range transport potential of PAHs had become a major concern as an environmental issue (Björseth 1979) when PAHs were first detected in the air and sediment of the North Polar Region (Lunde and Björseth 1977). The atmospheric long-range transport was suggested as a possible pathway of the occurrence of these compounds far from their sources. Recently, as the development of numerical simulation with computers, their long-range transport can be performed quantitatively.

Several critical processes are undergoing during their long-range transport. For instance, researches have shown that the reactions of NO<sub>2</sub> and OH radicals with PAHs are major processes for PAH degradation in the air (Brubaker and Hites 1998; Lammel et al. 2009). Further study revealed that the reactivity of PAHs in the gas phase is significantly larger than that in the particulate phase, and compared to NO<sub>2</sub>, reaction with OH radicals will be the dominant loss process of PAHs (Esteve et al. 2006). Through the comparison of four different gas–particle partitioning schemes, researchers came to the conclusion that a dual black carbon adsorption and organic matter absorption scheme could best describe PAH distributions and thus was suggested as an optimal scheme for long-range transport modeling (Lohmann and Lammel 2004). The suitability of the dual sorption scheme was further proven by model experiments conducted by Sehili and Lammel (2007) when the long-range transport of PAH emissions from Europe and Russia was studied. They also found that the soil compartment contributes a large fraction of the total environmental burden of PAHs. In their following study, the global atmospheric distribution and long-range transport of three PAH compounds were simulated using a global emission inventory (Lammel et al. 2009). This study indicated that gas–particle partitioning drastically influences the atmospheric cycling of PAHs, and that the degradation in the particulate phase must be slower than that in the gas phase. They found that the dual sorption scheme agrees best with the observations at remote sites and suggested that PAHs adsorbed in the particulate matter is shielded from the gas phase.

With consideration of major environmental behaviors such as gas–particle partitioning, OH degradation in the air, and surface–air exchange, Zhang et al. (2011) studied the trans-Pacific transport of PAHs emitted from Asia. It was found that the trans-Pacific transport flux was 1.6 times higher in the winter than in the summer, near ground concentration of BaP induced by Asia Sources in North America varied

between 1 and 20  $\text{pg/m}^3$ . The study also indicated a positive correlation between the interannual variability of transport and the Southern Oscillation Index. Based on a global emission inventory and the GEOS-Chem model, Friedman and Selin (2012) simulated long-range transport of three PAH compounds. Their model considered the dual sorption scheme of gas-particle partitioning with temperature-dependency and incorporated snow/ice scavenging and on-particle oxidation. The model provided a good agreement with observations at remote sites including sites in the high Arctic. Their next study evaluated impacts of climate change and emissions on the atmospheric PAH transport to the Arctic (Friedman et al. 2014) and revealed that emissions have a greater impact on multitude concentrations than climate does. The model also indicated a future “climate penalty” for volatile PAHs as a result of increasing temperature and corresponding surface-to-air fluxes of previously deposited PAHs and a “climate benefit” for particle-bound PAHs as a result of increasing deposition.

From the study on a single environmental process to numerical simulation of global transport, an integrated framework for PAH transport modelling has been gradually formulated (see, Fig. 2.2). Although many details remain to be explored, the framework can largely provide PAH transport simulation and environmental distributions with reasonable uncertainty. To achieve the simulation, a spatially resolved emission inventory is required. For global transport models, the spatial resolutions of emissions normally range from 1 to 5° latitude and longitude. Additionally, the critical environmental processes including photochemical degradation, wet/dry deposition, gas-particle partition should be considered. The reaction between gas phase PAHs and OH radicals is the major photochemical degradation process. The wet/dry deposition of particle-bound PAHs can follow the same scheme as black carbon and organic carbon, and the gas-particle partitioning scheme can follow the dual sorption scheme as mentioned above. Finally, the surface-air exchange should be included with consideration of the fates of PAHs in soil and seawater.



**Fig. 2.2** Major environmental processes and modeling framework for PAH compounds



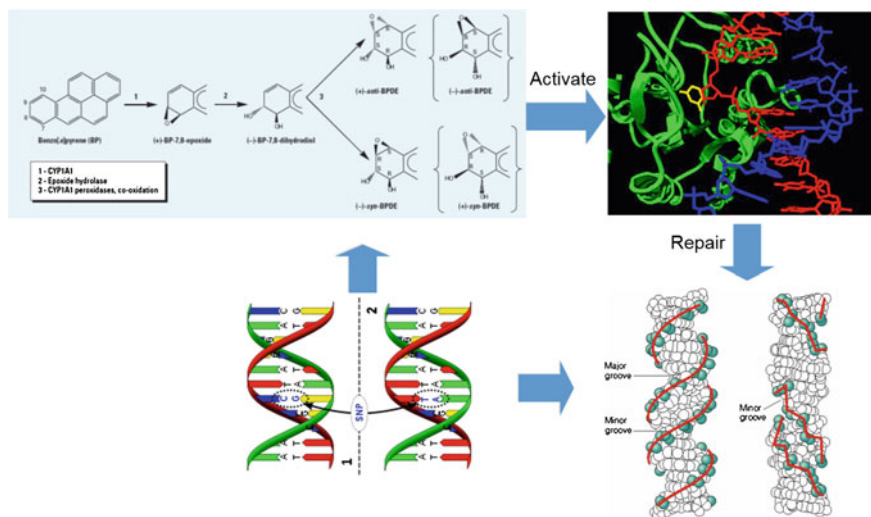
Although most global chemical transport models haven't involved PAH compounds in their original simulation, the simulation of these compounds can be achieved by adding emissions and necessary processes into the models, given that most of their environmental behaviors have been well studied and parameterized. Observations at remote sites are commonly chosen to evaluate model performance on the long-range transport capacity of PAHs, since comparisons at these sites partly avoid the influence of possible spatial bias induced by emission inventory. The available remote sites include the Alert site in the high Arctic, several sites monitored under the European Monitoring and Evaluation Program, and the measurements around the Great Lakes in the North America (EMEP 2012; Wang et al. 2010; Friedman and Selin 2012).

## 2.4 Toxicity of Polycyclic Aromatic Hydrocarbons

The greatest toxic concern of PAH compounds came from the observations that some of these compounds may cause tumors in humans (Boström et al. 2002). The first relevant event was reported in 1775 when scrotal cancer in chimney sweeps was observed and considered to originate from occupational exposure to soot (Pott 1775). One century later, elevated incidences of skin cancers were confirmed in workers in the coal tar industry (von Volkman 1875). In the early 1900s, soot, coal tar, and pitch were widely recognized to be carcinogenic to humans (Dipple 1985). In the following several decades, many epidemiologic studies and animal experiments have shown the carcinogenicity of these materials are associated with the fraction of PAH contents (Boström et al. 2002; Armstrong et al. 2004). Being evaluated by the International Agency for Research on Cancer, the carcinogenicity of PAHs in humans seems to be beyond dispute (IARC 1984a, b, 1985, 1987, 1989), and one of the high molecular weight PAHs, benzo[a]pyrene, has been recently classified into the group of the most carcinogenic agents (IARC 2014).

Among all the exposure routes, inhalation exposure to ambient PAHs and the related lung cancer risks are the most concerned aspect. Armstrong et al. (2004) conducted a review and meta-analysis of published reports of thirty-nine occupational epidemiologic studies. Their study showed the average estimated unit relative risk at 100  $\mu\text{g}/\text{m}^3$  years benzo[a]pyrene to be 1.20 (1.11–1.29 as 95 % confidence interval). Several studies have shown that benzo[a]pyrene can be regarded as a good indicator for risk assessment of inhalation exposure to PAH mixtures in the air (Boström et al. 2002; Muller 1997).

Carcinogenesis is believed to be a multistep, multimechanism process involving mutagenic events, epigenetic events, and altered cell survival (Boström et al. 2002; Hanahan and Weinberg 2000), and the carcinogenic process is often divided into three steps, including initiation, promotion, and progression (Pitot and Dragan 1996). PAH compounds may act at different steps in the carcinogenic process and may exert both mutagenic and epigenetic actions (Boström et al. 2002). One of the



**Fig. 2.3** A potential carcinogenic process. Through metabolic activation, PAHs can be converted into reactive electrophilic intermediates. The intermediates can then form adducts with DNA which influence transcription and protein synthesis. The adducts can be repaired by DNA repair enzyme

most important properties of PAHs is their metabolic conversion to reactive electrophilic intermediates (Boström et al. 2002). These intermediates can not only form adducts with DNA which induce mutations and eventually tumors, but also react with other cellular targets and interfere with transcription, DNA replication, and protein synthesis (Boström et al. 2002; Sims and Grover 1974; Thakker et al. 1985) (Fig. 2.3). Other properties of PAHs that are associated with routes of action include the high affinity to the cytosolic aryl hydrocarbon and the inhibitory effect on gap junctional intercellular communication (Boström et al. 2002).

However, researches on animal experiments and occupational environments may hinder the underlying mechanisms in ambient environments, since ambient PAH levels are often much lower. Evidences from animal experiments illustrate a non-linear dose-response relationship that show a significant increase of response in high exposure levels (Ehrenberg and Scalia-Tomba 1991), but this relationship may not be suitable for ambient exposure. The reason may rely on the two steps in the carcinogenic process at which PAHs can act. In the first step, namely the initiation, the dose-response relationship is regarded as a linear relation, while in the second step as the promotion, the relationship is nonlinear. In the ambient environment, the PAH action may not penetrate to the promotion step, and thus, a linear dose-response function can better describe the relationship.

## 2.5 Inhalation Exposure and Risk Assessment of Polycyclic Aromatic Hydrocarbons

PAH exposure routes that are associated with health effects involve oral exposure, inhalation exposure, and dermal exposure. Inhalation exposure is tightly linked with ambient atmospheric PAH contamination. As mentioned above, a linear function can represent the dose–response relationship in the ambient environment.

Two different methods can be applied to assess health risk of PAHs due to inhalation exposure in the ambient air. The first one is to sum up the risks from individual PAH compounds based on the dose–response relationship determined from animal experiments. Generally, a nonlinear function for a selected compound can be obtained between exposure dose and response in the experiments. A cancer slope factor of the compound is then calculated as the slope of the reduced linear risk function in the ambient level of exposure dose. The cancer risk of the specific compound can be addressed using the cancer slope factor and exposure dose. Additionally, toxic equivalency factors (TEFs) can be generated from the comparison of different compounds (see, Table 2.3). TEFs describe the relative toxicity of different compounds under the same exposure dose. Hence, given the risk of a single compounds and a compound profile in the mixture, risks of every individual compounds can be calculated directly using the TEF values, and the risk of the mixture is evaluated by summing-up risks of individual compounds.

The other approach is to use BaP as an indicator of the whole PAH mixture and assess the risk based on the dose–response relationship from epidemiologic studies. Unit Risk is often used to achieve risk assessment. The cancer risk from exposure to

**Table 2.3** Toxic equivalency factors of sixteen PAHs

Compound	TEF
NAP	0.001
ACY	0.001
ACE	0.001
FLO	0.001
PHE	0.001
ANT	0.01
FLA	0.001
PYR	0.001
BaA	0.1
CHR	0.01
BbF	0.1
BkF	0.1
BaP	1
IcdP	0.1
DahA	1
BghiP	0.01

ambient PAH mixture can be calculated as the product of the Unit Risk and the long-term average BaP concentration. Since most epidemiologic studies that focused on health impacts of PAHs took place in occupational environments such as coking plants and aluminum smelter workplaces, application of this approach has to assume first the similarity of compound profiles in the ambient air and in workplaces where the epidemiologic studies were carried out (Boström et al. 2002). This assumption considerably introduces uncertainty into risk assessment. However, this approach has still been chosen by the World Health Organization for risk assessment in the Air Quality Guidelines for Europe (WHO 2000). Because compared to this approach (using BaP as an indicator), the first approach (the summing-up approach) underestimates the risk by about two orders of magnitude (Muller 1997). In fact, only a handful of PAH compounds can be taken into consideration for summing up, which leads to significant underestimate.

Evidence from molecular epidemiologic studies has indicated that individual susceptibility plays an important role in cancer development in humans under environmental stress (Perera 1997). However, it is a challenge to distinguish the relative contributions of the exposure dose and individual susceptibility to the risks associated with PAHs (Perera 1997). A previous attempt has been made in a study in which the risks of PAH exposure inducing lung cancer were assessed with consideration of susceptibility associated with individual genetic polymorphisms in the Chinese population (Zhang et al. 2009). The results showed that the lung cancer susceptibility of half of the Chinese population varied from 35 to 280 % of the average, and the risk of 5 % of the most susceptible Chinese population is at least 10 times higher than that of the average population. The differences in individual susceptibility can be caused by gender, age, genetic heritage, etc. Their influences on risk assessment of PAH exposure will be discussed in this thesis. Consideration of these factors can reduce the uncertainty in risk assessment and provide more information for policy making.

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