PART 1

ACTIVATED CARBON ADSORPTION TECHNOLOGIES

For drinking water treatment, activated carbon is primarily used in two ways: fixed bed reactors with granular activated carbon (GAC) and slurry application of the smaller powdered activated carbon (PAC). In general, the manufacturing process is similar for both products. PAC is applied in a manner such that it cannot be economically recovered from the water treatment stream, so it is removed with the process solid residuals. For surface water treatment, GAC can be applied as media in a filter, termed a *filter adsorber*, or applied in a reactor positioned after the filter, a post-filter contactor. For direct groundwater treatment, reactors similar to post-filter contactors are used. After the adsorption capacity of GAC is exhausted, it must be removed from the fixed-bed reactor and reactivated either on-site or off-site.

While the physical and chemical characteristics of activated carbon can give some insight into the potential adsorption behavior, actual adsorption tests with the specific activated carbon and the compound of interest in the actual water yield the most useful information. These tests are described in the next chapters and include: (a) batch equilibrium capacity tests, i.e., "isotherms"; and (b) methods that assess both capacity and kinetics, including PAC jar tests and GAC bench-scale and pilot-scale column tests.

One critical factor that affects the adsorption efficiency is the influence of the background matrix on the adsorption capacity of the target compound, whether it is naturally occurring compounds like the odorants 2-methylisoborneol (MIB) and geosmin, or anthropogenic compounds like trichloroethylene and atrazine. Increases in the concentration of the compounds that compose the background matrix decrease the adsorption capacity for the target compound. Activated carbon can also be used to control the formation of disinfection by-products (DBPs) through the adsorption of dissolved organic matter (DOM). The critical factor here is understanding the effect of the influent DOM concentration on the adsorption efficiency.

The objective of part 1 of this book is to provide an understanding of the fundamentals of adsorption by activated carbon from a practical perspective. The chapters include adsorption fundamentals (chapter 1), the basics of PAC systems (chapter 2), and the basics of GAC systems (chapter 3). The approach taken herein is a semi-quantitative one that intends to educate the reader on the factors that affect activated carbon adsorption. The approach is based on well-established principles, and the reader is referred to several existing monographs on activated carbon adsorption that provide the extensive literature review and the quantitative basis that underline the approach taken herein (e.g., Sontheimer et al. 1988 and Summers et al. 2010).

References

Summers, R.S., D.R.U. Knappe, and V.L. Snoeyink. 2010. Adsorption of organic compounds by activated carbon. In *Water quality and treatment*, 6th ed. J.K. Edzwald, ed. New York: McGraw-Hill and American Water Works Association.

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Chapter 1

Fundamentals of Adsorption

Adsorption is used in water treatment to remove dissolved contaminants by physical phase separation in either a steady-state (PAC) or a nonsteady-state (GAC) process. Adsorbates accumulate on the surface of the adsorbent until the adsorption capacity is consumed. In the PAC process, the PAC is removed from the water treatment stream as a process solids residual. In the GAC process, once the effluent concentration of the adsorbate reaches the treatment objective, the GAC must be replaced in order to continue satisfactory performance. The high internal surface area of activated carbon provides enough adsorption capacity to make it an economical choice of adsorbent. However, regardless of how much surface area is provided for adsorption, different target compounds have various affinities for the surface phase versus the water phase, making adsorption dependent on the properties of the target compound and the background matrix.

This chapter first provides an introduction to the process of manufacturing activated carbon. Then the physical-chemical process fundamentals of adsorption systems are discussed.

Activated Carbon Manufacturing and Properties

Activated carbons are manufactured from a wide range of carbonaceous materials. The most common precursor materials for use in drinking water treatment are bituminous coal, lignite coal, coconut shells, and wood. The choice of raw material affects the internal pore structure, surface area distribution, and surface chemistry. Commercial activated carbon production is a two-step process of carbonization of the precursor materials and activation. In the pyrolytic carbonization process, the temperature of the raw material is raised to the range of 500 to 800 °C in the absence of oxygen. Volatile organic matter of the raw material is thermally released, and the carbon atoms realign to form a more crystalline structure. The carbonized product at this point in the process is heavily influenced by the raw materials used. For activated carbon products used in water treatment, a thermal or physical activation process then follows in which the temperature of the carbonized product is increased to the range of 850 to 1,000 °C in the presence of an oxidizing agent, typically steam or carbon dioxide. Activation increases the pore sizes and creates a continuous pore structure, which increases the micropore volume (pore width < 2 nm) and the internal surface area. The mesopore and macropore volumes (pore width > 2 nm) are critical to the internal transport of the compound to the micropore surface area, which is where most of the adsorption occurs. Activated carbon can be produced in a

Table 1-1.   Typical activated carbon physical characteristics

Porosity	Surface area (m²/g)	Particle density (kg/m³)	Apparent dry density (kg/m³)		Average particle diameter (mm)	
			GAC	PAC	GAC	PAC
0.5–0.8	500–1,200	600–850	300–650	200–750	0.6–3.0	0.01–0.03

* 1 lb/ft³ = 16 kg/m³

direct activation process in which the raw material is crushed and then activated, or in a reagglomeration process in which the raw material is crushed, reagglomerated, crushed again, and then activated. Table 1-1 lists some typical physical properties of commercial activated carbon used in water treatment.

The apparent density of GAC is the as-received density of unstratified dry material. The bed density is the density after the GAC is placed in the filter or contactor, backwashed, and drained. Bed density is generally about 90 percent of the apparent density due to stratification.

Once the adsorption capacity of GAC is exhausted, it is removed from the filter or contactor and the adsorption capacity is restored in a reactivation process. The reactivation requires conditions very similar to the activation process used in the original production. During reactivation, some adsorbed compounds are thermally desorbed, but most compounds, including the adsorbed DOM, are oxidized on the GAC surface.

The surface chemistry of activated carbon is a product of the raw materials and the activation process. The noncarbon content of activated carbon varies but is typically in the 10 to 20 percent range and is mostly made up of surface oxygen functional groups and minerals (e.g., ash), both of which can negatively affect the adsorption of some compounds.

Adsorption

Adsorption is the accumulation of a substance (the adsorbate) on the surface of a solid (the adsorbent). While most, but not all, organic compounds of interest in drinking water quality are adsorbable by activated carbon, it is the degree of adsorbability that is important, as it dictates the cost of using activated carbon. For example, DOM consists of a mixture of compounds of which about 90 percent are adsorbable to some degree. Large-molecular-weight compounds are excluded from internal adsorption sites, and hydrophilic DOM compounds of any size are not attracted to the adsorbent surface. The most strongly adsorbed DOM components are low- and medium-molecular-weight hydrophobic compounds.

Adsorption of compounds of interest in drinking water by activated carbon occurs mostly by nonspecific physical adsorption as opposed to site-specific chemical adsorption. Physical adsorption is caused by nonspecific secondary binding mechanisms, in which electrons are shared between the adsorbate and the adsorbent rather than electrons being transferred, as occurs in chemical adsorption. Physical adsorption can be reversed in response to a change in the concentration gradient between the bulk liquid phase and the adsorbed solid phase. Because activated carbon has a

nonhomogeneous surface with specific function groups, a limited degree of chemical adsorption for some compounds is possible.

Batch Adsorption Kinetics

As illustrated in Figure 1-1, removal of a compound by activated carbon is a three-step process: external mass transfer (EMT), internal mass transfer (IMT), and adsorption. The first step is external mass transfer of the compound from the bulk liquid phase through the hydrodynamic layer that surrounds the activated carbon particle. External mass transfer, or film diffusion, can be important and control the overall kinetics of the adsorbate uptake when the flow through a GAC bed is very low or when the mixing in a PAC reactor, like the flocculation basin, is low. After external mass transfer delivers the adsorbate to the external surface of the activated carbon particle, internal mass transfer conveys the adsorbate to an adsorption site on the surface of the activated carbon where adsorption occurs. The adsorption reaction of the compound at the activated carbon surface is very fast and is not considered to be rate limiting. Thus, the overall uptake is thought of as a two-step mass transfer process

Figure 1-1.   Illustration of the mass transfer–adsorption process

of external and internal mass transfer. Because external diffusion and internal diffusion occur in series, the slowest step is rate limiting. Both external and internal mass transfers are diffusion-based phenomena where the driving force is the concentration gradient of the adsorbate: the difference between the bulk concentration, C_b, and the internal concentration at equilibrium, C_e. Internal mass transfer is often the slowest and thus controls the overall uptake of an adsorbate. Internal mass transfer may occur through the pore fluid or along the pore wall in parallel processes. Both pore diffusion and surface diffusion are controlled by both the activated carbon pore structure and the adsorbate properties. Because the processes occur in parallel, the faster step, which is typically pore diffusion under most conditions relevant to water treatment, controls the overall uptake of an adsorbate.

The kinetics of the overall adsorbate uptake can be illustrated using a completely mixed batch reactor (CMBR) such as shown in Figure 1-2, where V is the reactor volume, M_{GAC} is the mass of activated carbon added, and C_0 is the initial liquid phase adsorbate concentration. Mixing in the reactor is important as the external mass transfer resistance can be reduced by increasing the mixing enough to reduce the thickness of the stagnant film surrounding the particle.
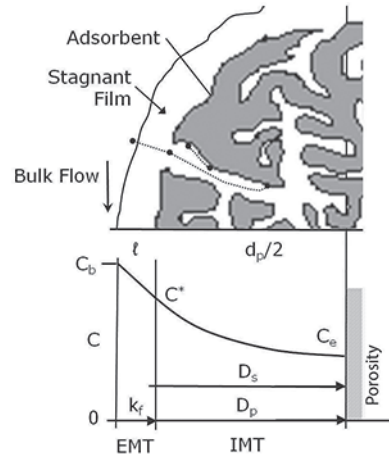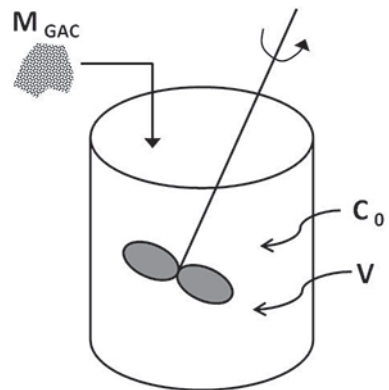

Figure 1-2.   Activated carbon adsorption in a completely mixed batch reactor
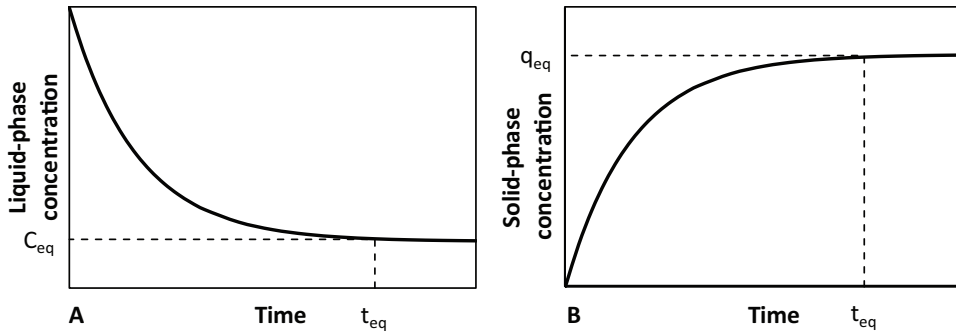
Figure 1-3. Adsorbate uptake kinetics in a CMBR shown as both (a) liquid-phase and (b) solid-phase concentrations

After the addition of the activated carbon, the liquid phase concentration of the adsorbate in the reactor will decrease as a function of time. Eventually, the net uptake of the adsorbate reaches zero. The adsorbate-adsorbent pair is now at equilibrium. The liquid phase concentration at this point is termed the *equilibrium liquid-phase concentration*, C_{eq}, and the time is termed the *time to equilibrium*, t_{eq}. Figure 1-3 illustrates the liquid-phase concentration history in a batch test along with the kinetics of the adsorbate concentration on the adsorbent surface, solid-phase concentration, q. The solid phase concentration at equilibrium, q_{eq}, is attained at the same time as C_{eq}.

As the adsorbent dose increases, the equilibrium liquid-phase concentration decreases as shown in Figure 1-4a. In addition to C_{eq} being affected by the dose increase, the time to equilibrium is also affected. Other factors that affect the kinetics or time to equilibrium, but not the equilibrium liquid-phase concentration, include the size of the adsorbent, expressed here as the diameter (d_{AC}); diffusion coefficient of the adsorbate (D); initial liquid phase concentration; and amount of shear or mixing imparted on the external surface of the adsorbent. The adsorbent size affects the internal mass transfer. As the adsorbent size decreases, the internal diffusion path length the adsorbate must travel to an adsorption site decreases, and the adsorption sites can be reached more quickly. The same mass of smaller adsorbent particles
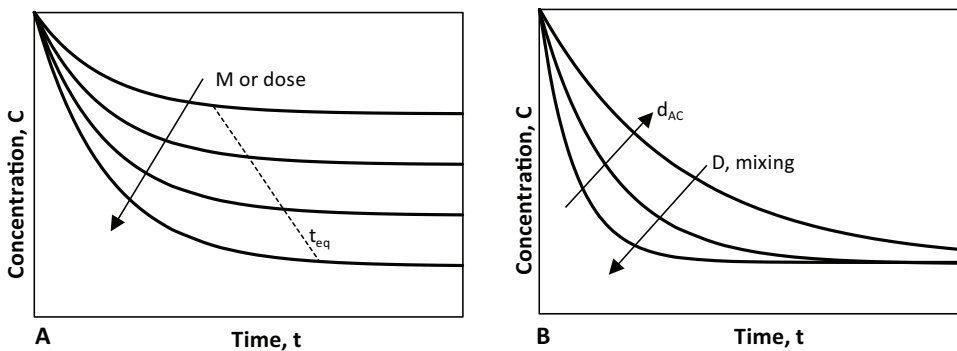

Figure 1-4. Adsorbate uptake kinetics and equilibrium liquid-phase concentration in a CMBR as affected by (a) the adsorbent dose and (b) adsorbent size (d_{AC}), adsorbate diffusivity (D), and system mixing or shear

achieves equilibrium state more rapidly, lower t_{eq}, than that of large adsorbent particles. As the diffusion coefficient and the concentration of the adsorbate increases, the flux into the adsorbent increases and the adsorption sites can be reached more quickly, also decreasing the time to equilibrium. Factors affecting time to equilibrium are summarized in Eq 1-1:

$$t_{eq} = f\left(d_{AC}^2,\ D^{-1},\ C_0^{-1}\right) \qquad\qquad \text{(Eq 1-1)}$$

When external mass transfer is an important factor, increasing the mixing in the CMBR or the fluid velocity decreases the boundary layer around the particle, thus increasing the overall uptake kinetics.

Adsorption Equilibrium, or Isotherm

Mass Balance

As was shown in Figure 1-4a, increasing the mass of activated carbon added to a batch reactor will decrease the equilibrium liquid-phase concentration, and the equilibrium solid-phase concentration increases by mass balance as shown in Eq 1-2:

$$q_{eq} = (C_0 - C_{eq}) \times V/m = (C_0 - C_{eq})/\text{dose} \qquad\qquad \text{(Eq 1-2)}$$

If an experiment is run that varies the adsorbent dose and/or initial concentration, the equilibrium solid-phase concentration can be calculated, and a relationship with the equilibrium liquid-phase concentration can be established. This adsorption equilibrium relationship, often termed *isotherm*, allows the adsorption capacity of the adsorbent to be calculated at any given liquid-phase concentration.

EXAMPLE 1-1. Different masses of activated carbon are added to bottles that contain an adsorbate of interest to yield the doses noted in Table 1-2. Two different initial concentrations of the adsorbate were used. To facilitate the mass transfer, small activated carbon sizes were used, and the bottles were shaken. Preliminary experiments determined the time needed to achieve equilibrium (Figure 1-3). After waiting the prescribed equilibrium time, activated carbon was filtered from the solution, and the adsorbent equilibrium liquid-phase concentration was measured. Calculate the equilibrium solid-phase concentration at each dose and plot as a function of the equilibrium liquid-phase concentration.

SOLUTION. Using the mass balance (Eq. 1-2), the equilibrium solid-phase concentration was calculated. The results are shown in Table 1-2 and Figure 1-5.

The shape of the isotherm plot is typical of the adsorption of target compounds by activated carbon from laboratory clean water. As can be seen for single-solute isotherms, the equilibrium relationship is not a function of the initial concentration.

Equilibrium Adsorption Isotherm Models

To be more useful, equilibrium isotherm relationships can be fit with model equations. The two most common are the Langmuir and Freundlich models. The Langmuir model is more fundamental, has conceptual endpoints as it asymptotically approaches a maximum solid-phase concentration at the very high liquid-phase concentration range, and has a linear equilibrium relationship at the very low–concentration ranges. However, experience has shown that the empirical Freundlich model (Eq 1-3), which

Table 1-2. Example isotherm results

Initial concentration, C_0 2.00 mg/L			Initial concentration, C_0 0.500 mg/L		
Dose (mg/L)	C_{eq} (mg/L)	q_{eq} (mg/g AC)	Dose (mg/L)	C_{eq} (mg/L)	q_{eq} (mg/g AC)
5	1.53	92.77	5	0.295	40.74
10	1.18	81.47	10	0.18	31.82
25	0.575	56.87	25	0.056	17.75
50	0.224	35.50	65	0.01	7.50
100	0.066	19.27	150	0.002	3.35
200	0.0175	9.92	335	0.0004	1.50
400	0.0044	4.97	610	0.00012	0.82
800	0.00111	2.50			
1,600	0.00028	1.25			
3,000	0.00008	0.67			

is a power function, works best. The Freundlich model can be derived theoretically from the Langmuir model assuming a distribution of adsorption site energies.

$$q_{eq} = K_F \, C_{eq}^{1/n} \qquad \text{(Eq 1-3)}$$

Where

K_F = Freundlich constant
$1/n$ = Freundlich exponent

To simplify the notation in this chapter, the equilibrium liquid-phase and solid-phase concentrations are expressed as shown in Eq 1-4:

$$q = K_F \, C^{1/n} \qquad \text{(Eq 1-4)}$$

To fit the Freundlich parameters to the data, the Freundlich equation can be linearized as shown in Eq 1-5:

$$\log q = \log K_F + 1/n \log C \qquad \text{(Eq 1-5)}$$

When the isotherm data are plotted on a log scale as shown in Figure 1-5b, the slope of the line is the $1/n$ value, and K_F is evaluated on the y-axis at a concentration of 1 mg/L. For data expressed in the units shown, the value of $1/n$ is 0.5 and K_F is 75 $(mg/g)/(mg/L)^{1/n}$. The value of K_F depends on the units of the isotherm data. For the same data as above, but if C is in units of µg/L instead of mg/L, the value of K_F is 2.37 $(mg/g)/(µg/L)^{1/n}$.

The value of the Freundlich constant (K_F) can be useful in gauging the difference in the adsorption capacity of a given activated carbon for a range of compounds or
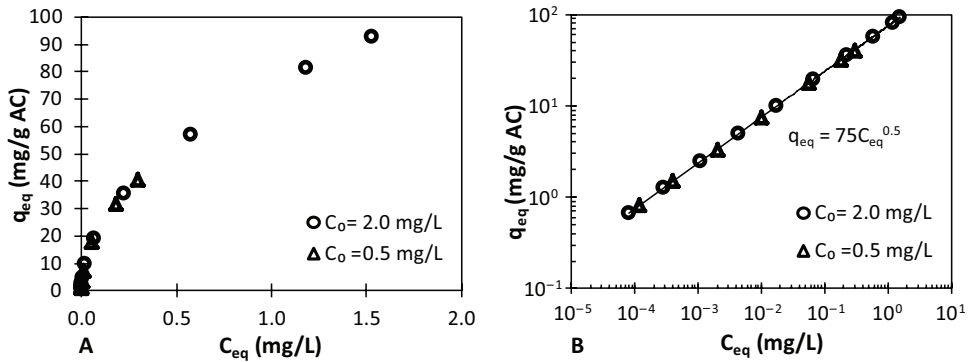
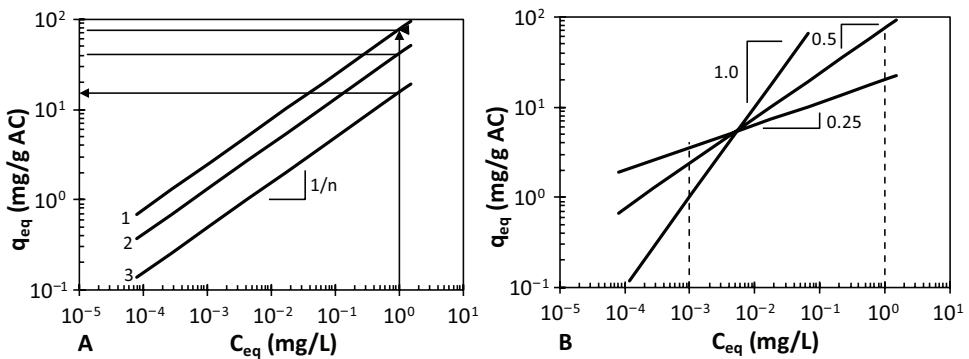**Figure 1-5.   Example isotherm (a) arithmetic plot and (b) logarithmic plot**

Figure 1-6.   Effect of Freundlich parameters (a) K_F and (b) 1/n on isotherm

the adsorption capacity of a given compound for a range of activated carbons, as long as the Freundlich exponent (1/n) value is relatively constant. Figure 1-6a illustrates three isotherms that all have a 1/n value of 0.50. The larger the K_F value, the higher the adsorption capacity for a given adsorbate-adsorbent pair.

However, when the 1/n values of the different isotherms are significantly different, as shown in Figure 1-6b, the K_F value is only useful in the concentration range around 1, in this case 1 mg/L. The isotherm plots converge at lower concentration ranges, and to compare adsorption capacities both the K_F and 1/n values are needed, or comparisons can be made on the basis of a solid-phase concentration at a given liquid-phase concentration (q_C). For example, $q_{0.001}$ can be use to represent the solid-phase concentration at a liquid-phase concentration of 0.001 mg/L (1 μg/L). For this example, the isotherm that yielded the lowest capacity in the mg/L range now has the highest capacity in the μg/L range.

Summers et al. (2010) have compiled a list of single-solute Freundlich parameters for more than 70 compounds of interest; the K_F values range from 100 to 0.1(mg/g)/(μg/L)$^{1/n}$, and the 1/n values range from approximately 0.8 to 0.1. In general, for the K_F values expressed in the units (mg/g)/(μg/L)$^{1/n}$ and in a concentration range around 1 μg/L, compounds with K_F values above 10 (mg/g)/(μg/L)$^{1/n}$ can be considered well removed by activated carbon, and compounds with K_F values below 1 (mg/g)/(μg/L)$^{1/n}$ can be considered poorly removed by activated carbon. These Freundlich parameter

values have been generated from isotherms with a single type of activated carbon, and the adsorption capacity for these compounds may be increased or decreased by the use of other activated carbon products.

A major limitation of the Freundlich model is that it should only be used to model the equilibrium data in the concentration range for which it was developed; it does not asymptotically approach a maximum adsorption capacity on the high concentration end, nor does it approach a linear isotherm on the low concentration range.

Competitive Adsorption

Caution must be exercised in not overusing single-solute Freundlich parameter values, as they are affected by the background matrix in the water to be treated. If other adsorbates are present in the water, as they are in nearly all practical cases, then the adsorption capacity for a specific compound decreases, even if the background contains only weaker adsorbing compounds. Figure 1-7 illustrates competitive adsorption behavior for a binary system in which Compound 1 is more strongly adsorbed than Compound 2 (i.e., $K_{F,1} > K_{F,2}$). In Figure 1-7a, the effect of the more weakly adsorbed Compound 2 on the adsorption of Compound 1 is shown. Increasing the initial concentration of Compound 2 causes the adsorption capacity of Compound 1 to decrease relative to the single-solute isotherm at $C_{2,0} = 0$. Figure 1-7b shows that increasing the initial concentration of more strongly adsorbing Compound 1 causes the adsorption capacity of Compound 2 to decrease relative to the single-solute isotherm at $C_{1,0} = 0$. As expected, the presence of the same amount of Compound 1 is more detrimental to the adsorption of Compound 2 (Figure 1-7b) than the other way around (Figure 1-7a). While not illustrated in Figure 1-7, the slope and shape of the isotherm in a mixture can also be affected.

Multisolute systems, like the adsorption of a target compound in the presence of DOM, add additional complexity to adsorption behavior. Competing adsorbates in the same molecular size range as the target adsorbate compete directly for adsorption sites (Li et al. 2003). Larger competing adsorbates may not be able to access micropores because of size exclusion. However, size exclusion may also result in partial or complete blockage of a pore. Partial blockage of a pore may slow kinetics of the target adsorbate, whereas complete blockage of a pore results in surface area behind the
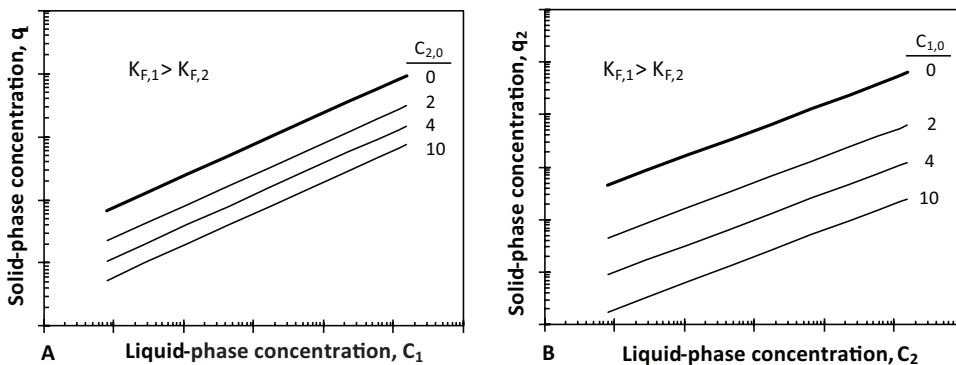

Figure 1-7.   Equilibrium isotherm in a binary mixture (a) strongly adsorbing Compound 1 and (b) weakly adsorbing Compound 2

blockage not being available for further adsorption. Because of its tendency to adsorb at multiple sites, adsorbed DOM is nondisplaceable to a large degree (Summers et al. 2010). Pore blockage and nondisplaceable adsorption cannot be described by ideal adsorbed solution theory (IAST). Therefore, reductions in adsorption capacity due to these mechanisms are often termed *fouling*.

DOM Equilibrium Adsorption Behavior

An understanding of DOM adsorption is important, as DOM serves as a precursor for the formation of DBPs. DOM is a heterogeneous mixture of organic compounds and normally dominated by hydrophobic organic acids such as humic and fulvic acids but also contains hydrophilic compounds that are weakly adsorbable or nonadsorbable. To measure DOM collectively, the total organic carbon (TOC) concentration is measured. Figure 1-8 illustrates the adsorption equilibrium relationship of DOM, as measured by TOC, at two initial DOM concentrations. The adsorption behavior of both isotherms displays a nonadsorbable fraction of about 10 percent, and as the equilibrium liquid-phase concentration increases, so does the adsorbability of the DOM in a nonlinear manner. The behavior shown in Figure 1-8 cannot be modeled by the Freundlich or Langmuir isotherms. When the initial DOM concentration decreases to a TOC = 2.5 mg/L, the adsorption capacity increases in all concentration ranges. The complex adsorption behavior of the DOM mixture limits the use of the isotherm in comparing the DOM adsorption behavior from different waters. Because the initial DOM concentration must be the same, it requires the use of complex adsorption models.
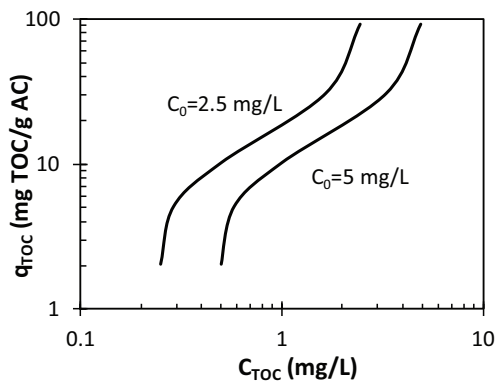

Figure 1-8.   Equilibrium adsorption isotherms of DOM at two different initial concentrations

Several other adsorbate properties can also affect the adsorption equilibrium relationship. Large-molecular-weight molecules (i.e., > 1,000 Daltons) are often larger in size than the micropores where much of the surface area is located; thus size exclusion can limit the amount adsorbed. Weak organic bases and acids, like humic and fulvic acids in DOM, can be affected by pH because adsorption capacity is higher in the neutral form than the ionized form.

Ideal Adsorption Reactors

The above adsorption fundamentals were developed for CMBRs that have no volumetric flow, Q, into the reactor. Reactors used in water treatment are flow-through reactors and behave in the envelope of performance of the two idealized flow-through reactors: the plug flow reactor (PFR) and the continuous flow stirred tank reactor (CFSTR), shown in Figure 1-9. The difference in behavior is critical in understanding activated carbon performance. These will be described in detail in the next two chapters, but an overview at this point is instructive.

The mixing required to suspend PAC media also destroys the spatial concentration gradient of the target compound, yielding CFSTR behavior. In the ideal CFSTR
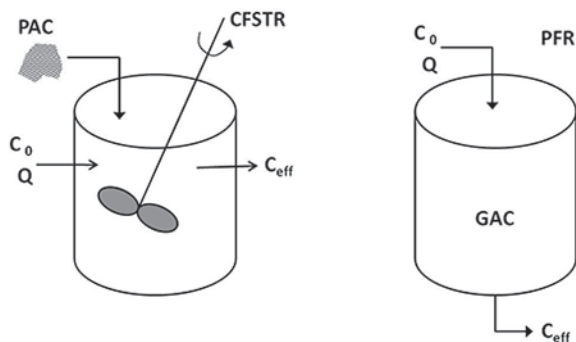
Figure 1-9.   Flow-through reactors: CFSTR for PAC and PFR for GAC

(Figure 1-9), the mixing conditions make the concentration of the target compound equal to the effluent concentration, C_{eff}, at all points in the reactor. On the other end of the spectrum, the generally laminar flow through a packed GAC bed maintains the concentration gradient from the reactor influent to the effluent and yields PFR behavior. With long enough run times, the GAC will come to equilibrium with the influent concentration, C_0.

The adsorbent in the CFSTR comes to equilibrium with the effluent concentration in Eq 1-4, whereas a PFR will come to equilibrium with the influent concentration.

EXAMPLE 1-2. Calculate the solid phase concentration, q_{eq}, of atrazine in laboratory clean water at equilibrium in a PFR and a CFSTR. The influent concentration of atrazine is 50 μg/L, K_F is 38.7 $(mg/g)(L/μg)^{1/n}$, and $1/n$ is 0.291. The treatment objective or desired effluent concentration for atrazine is the maximum contaminant level (MCL) of 3 μg/L.

SOLUTION. The solid-phase concentration is calculated directly from Eq 1-3 or 1-4. The PFR comes to equilibrium with the influent concentration of 50 μg/L:

$$q_{eq\,PFR} = K_F \times C_0^{1/n} = 38.7\ \frac{mg}{g} \times \left(\frac{L}{\mu g}\right)^{1/n} \times \left[50\ \frac{\mu g}{L}\right]^{0.291} = 121\ \frac{mg\ atrazine}{g_{AC}}$$

The CFSTR comes to equilibrium with the effluent concentration, in this case 3 μg/L:

$$q_{eqCFSTR} = K_F \times C_e^{1/n} = 38.7\ \frac{mg}{g} \times \left(\frac{L}{\mu g}\right)^{1/n} \times \left[3\ \frac{\mu g}{L}\right]^{0.291} = 53.3\ \frac{mg\ atrazine}{g_{AC}}$$

Thus the GAC in the PFR adsorbs over twice the amount of atrazine per mass of activated carbon compared to the PAC in a CFSTR. The PFR is a more efficient reactor.

Another way of comparing the efficiency of the PAC in a CFSTR to GAC in a PFR is the use rate. Table 1-3 defines the use rate in two different forms for each reactor: mass of AC per volume of water treated or mass of AC used per day. As such, lower use rates are more favorable. Note that the ideal PFR removes all the contaminant until breakthrough is reached; therefore, the effluent concentration is zero.

These calculated use rates are representative of single-solute systems (no background natural organic matter [NOM]). Typical use rates in drinking water treatment may be more than an order of magnitude higher.

EXAMPLE 1-3. For the problem given in Example 1-2, calculate use rate in kg/1,000 m³ and in kg/day if the flow rate is 4.0×10^4 m³/day (10.6 mgd).

SOLUTION. See Table 1-3. The GAC system requires less than half the amount of AC compared to the PAC system. *Note:* in practical use, the adsorption capacity for

Table 1-3. Use rate definitions using isotherm parameters and Example 1-3 calculations

Use Rate	PAC–CFSTR	GAC–PFR
Mass of AC/volume water treated	$(C_0 - C_{eff})/q_{eqCFSTR}$ $= (C_0 - C_{eff})/K_F \times C_{eff}^{1/n}$	$(C_0 - C_{eff})/q_{eq\,PFR}$ $= (C_0 - C_{eff})/K_F \times C_0^{1/n}$
Mass of AC/day	$Q \times (C_0 - C_{eff})/ q_{eq\,CFSTR}$ $= Q \times (C_0 - C_{eff})/ K_F \times C_{eff}^{1/n}$	$Q \times (C_0 - C_{eff})/ q_{eq\,PFR}$ $= Q \times (C_0 - C_{eff})/ K_F \times C_0^{1/n}$

Example 1-3		
Use Rate	PAC–CFSTR	GAC–PFR
Mass of AC/volume water treated	$= (50\ \mu g/L - 3\ \mu g/L) /$ $53.3\ \mu g$ atrazine/mg AC $= 0.88$ mg AC/L $= 0.88$ g AC/m³ $= 0.88$ kg AC/1,000 m³ $= 0.0074$ lb/1,000 gal	$= (50\ \mu g/L - 0\ \mu g/L) /$ $121\ \mu g$ atrazine/ mg AC $= 0.41$ mg AC/L $= 0.41$ g AC/m³ $= 0.41$ kg AC/1,000 m³ $= 0.0035$ lb/1,000 gal
Mass of AC/day	$= Q \times (50\ \mu g/L - 3\ \mu g/L) /$ $53.3\ \mu g$ atrazine/mg AC $= 4.0 \times 10^4$ m³/day $\times$ 0.88 g AC/m³ $= 35$ kg/day $= 78$ lb/day	$= Q \times (50\ \mu g/L - 0\ \mu g/L) /$ $121\ \mu g$ atrazine/mg AC $= 4.0 \times 10^4$ m³/day $\times$ 0.41 g AC/m³ $= 16$ kg/day $= 36$ lb/day

the target compound is much lower due to the competitive effects from background organic matter, resulting in higher use rates.

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