Environ. Sci. Technol. 1982, 16, 274–278

Correlation of Bioconcentration Factors

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■ The physical–chemical factors influencing bioconcentration of organic solutes are examined, and it is suggested that a directly proportional relationship exists between bioconcentration factor and octanol–water partition coefficient. A one-constant correlation is derived and shown to give a satisfactory fit of the available data for fish. The correlation can be extended to give a simple relationship between bioconcentration factor and aqueous solubility.

Introduction

A very important aspect of the assessment of the environmental fate of chemicals is the prediction of the extent to which these substances will achieve concentrations in biotic phases (such as fish), which may be orders of magnitude greater than the concentrations in the media (such as water) in which they reside. A convenient characterizing quantity is the bioconcentration factor K_B, which is the dimensionless ratio of the concentration in the biota to that in the medium as described, for example, by Neely (*1*). The bioconcentration process is often viewed as a balance between two kinetic processes, uptake and depuration, as quantitifed by first-order rate constants K_1 and K_2, respectively, the resultant differential equation being

$$dC_B/dt = K_1C_M - K_2C_B \quad (0)$$

where C_B is the biotic concentration, C_M is the medium concentration (in identical units of mg/L or mol/m^3) and t is time. Integration from an initial condition of C_B and t equal to zero gives

$$C_B = (K_1/K_2)C_M(1 - \exp(-K_2t)) \quad (1)$$

At long times C_B tends toward $(K_1/K_2)C_M$, or K_BC_M, and K_B is then the ratio of the uptake and depuration rate constants. Values of K_B can be obtained from long-term exposure measurements (to give C_B) or by kinetic measurements as described by Branson (*2*).

The important point is that C_B rises in an approximately first-order form to approach K_BC_M.

An alternative approach is to view the biotic phase (e.g., fish) as an inanimate volume of material that is approaching thermodynamic equilibrium with its medium (e.g., water) as defined by the chemical potential or fugacity of the bioconcentrating-persistent solute. In this case the differential equation can be reformulated to give

$$dC_B/dt = K_3(C_M - C_B/K_B) \quad (2)$$

which integrates to give

$$C_B = C_MK_B(1 - \exp(-K_3t/K_B)) \quad (3)$$

which is also a first-order process in which C_B approaches K_BC_M at long exposure times.

Uptake rate measurements are thus inherently incapable of discriminating between these differential equations. Undoubtedly the actual uptake process is more complex, being a combination of uptake from the medium and from food, which may also be at, or approaching, equilibrium with the medium.

The latter approach has the attractive feature that K_B can be related to physical–chemical equilibrium properties. This approach will be invalid when the solute is appreciably degraded by chemical or biochemical processes in the biotic phase; however, the cases of greatest environmental interest tend to be those in which the solute is nonreactive or recalcitrant.

Some support for the equilibrium approach can be found in the observation by Neely (*3*) and later Veith et al. (*4*) that K_B is well correlated with a truly equilibrium quantity K_{OW}, the octanol–water partition coefficient. Clayton et al. (*5*) have also suggested that near-equilibrium conditions may exist in biotic phases, especially when their differing internal composition is taken into account.

Kenaga (*6*, *7*) and Kenaga and Goring (*8*) have comprehensively reviewed K_B data and have developed correlations with K_{OW} and aqueous solubility. Baughman and Paris (*9*) have critically reviewed microbial bioconcentration and have shown that the data are consistent with passive equilibrium partitioning.

Derivation

If we view an organism, such as a fish, as consisting of a number of phases of differing chemical composition and of volume fraction y_i, then at thermodynamic equilibrium the chemical potential or fugacity (f) of the bioconcentrating solute will be equal in the medium (f_M) and in each phase (f_i), i.e.

$$f_M = f_1 = f_2 = f_3 = f_i$$

Now in each phase f_i can be related to the concentration C_i expressed as the mole fraction x_i, the phase molar volume v_i, the activity coefficient γ_i, and the reference fugacity f_R (on a Raoult's law convention) as

$$f = x_i\gamma_if_R = C_iv_i\gamma_if_R \quad (4)$$

It is assumed here that x_i is small, thus the presence of the solute has a negligible effect on the phase molar volume v_i.

The amount in each phase m_i (moles) is thus C_iy_iV, where V is the organism volume, and the average biotic concentration C_B becomes

$$C_B = \Sigma m_i/V = \Sigma C_iy_i = (f/f_R)\Sigma(y_i/\gamma_iv_i) \quad (5)$$

Now the fugacity in the medium phase will be equal and will be expressed by

$$f = x_M\gamma_Mf_R$$

Further, the medium concentration C_M is x_M/v_M at low concentrations, thus f is $C_Mv_M\gamma_Mf_R$ or C_M is $f/(v_M\gamma_Mf_R)$. It follows that K_B is given by

$$K_B = C_B/C_M = \gamma_Mv_M\Sigma(y_i/\gamma_iv_i) \quad (6)$$

In the case of water as the medium a convenient correlating quantity is the octanol–water partition coefficient K_{OW}, which is the ratio of the solute octanol concentration C_O to that in water C_M in which

$$K_{OW} = C_O/C_M = (x_O/v_O)/(x_M/v_M) \quad (7)$$

but the fugacities are equal, i.e.

$$f = x_O\gamma_Of_R = x_M\gamma_Mf_R$$

thus

$$K_{OW} = \gamma_Mv_M/\gamma_Ov_O \quad (8)$$

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Table I. Values of the Constants n and b in the Equation $\log K_B = n \log K_{OW} + b$

n	b	ref	n	b	ref
0.907	−0.361	9	0.85	−0.70	4
0.837	−0.770	9 (from *11*)	0.542	+0.124	3
1.160	−0.75	*12*			

where subscript O refers to the octanol phase.

It is thus not surprising that for a variety of compounds of differing values of K_{OW} (or γ_M), K_B correlates well with K_{OW} since both are largely controlled by the quantity γ_M, the activity coefficient of the solute in the medium—here water. This analysis suggests that the ratio of K_B to K_{OW} may show some constancy since

$$K_B/K_{OW} = \sum(y_i/\gamma_i v_i)\gamma_O v_O \quad (9)$$

The implication is that for organisms of similar composition (y_i) with phases of similar properties (γ_i) in which the ratio of the significant phase activity coefficients (γ_i) to that of octanol (γ_O) is fairly constant, the ratio of K_B to K_{OW} should be fairly constant. A suitable test would be to tabulate K_B/K_{OW} for an organism for compounds of widely varying K_{OW} or K_B.

In practice for hydrophobic organic solutes it is likely that the lipid phase is the dominant phase of bioconcentration because of its relatively low γ_i value, in which case if the other phases are ignored

$$K_B/K_{OW} = y_L\gamma_O v_O/\gamma_L v_L \quad (10)$$

where subscript L refers to the lipid phase. Organisms of high lipid content (y_L) should exhibit high K_B values, an observation confirmed experimentally by Clayton et al. (*5*).

Any variation in K_B/K_{OW} between solutes will arise from changes to γ_O/γ_L, which can be viewed as the ratio of the solute solubilities in the lipid and octanol phases. Octanol is regarded as a satisfactory surrogate model compound for lipid phases containing a comparable balance between hydrophobicity and hydrophilicity or polar character, as is evident from its very successful use in structure–activity relationships (*10*). Although γ_O/γ_L may change with, for example, increasing molecular weight, it seems unlikely that the change will be large. This suggests testing the one-constant (A) correlation

$$K_B/K_{OW} = A \text{ or } \log K_B = \log K_{OW} + \log A \quad (11)$$

Traditionally, correlations of K_B and K_{OW} have taken the form

$$\log K_B = n \log K_{OW} + b \quad (12)$$

Values of n and b are given in Table I for various biota. It is striking that n is consistently close to unity (as is expected) but is usually lower in value. Possible reasons for this are discussed later.

It can be argued that although there is no fundamental basis for suggesting a linear correlation of $\log K_B$ and $\log K_{OW}$, such a correlation is an obvious first approach since K_B and K_{OW} vary by a factor of 10^4–10^6, thus direct regression of K_B with K_{OW} distorts the analysis. Further, errors in K_{OW} and K_B tend to be constant factors (e.g., factor of 2) rather than absolute amounts, thus it is appropriate to use a least-squares regression of the logarithmic quantities. It is as valid to correlate K_B/K_{OW} with molecular properties and in essence the eq 12 type correlation can be written in this form as

$$K_B/K_{OW} = 10^{(b-(1-n)\log K_{OW})} \quad (13)$$

Clearly, when n is unity, the right-hand side becomes a constant. It is noteworthy that the Veith correlation with n equal to 0.85 gives a low coefficient of 0.15 on $\log K_{OW}$, thus the right hand varies only by a factor of 4 when K_{OW} changes from 10^2 to 10^6. Since the accuracy of K_B and K_{OW} measurements is often suspect, it is apparent that it is difficult to determine n accurately. It is striking that the values of n in Table I vary around unity, the average value being 0.86.

It is thus interesting to test the one-constant equation (11) against the two-constant equation (12), and there are obvious incentives to use the former unless the fit is demonstrably poorer.

Data Analysis

Veith et al. (*4*) have measured or compiled K_B and K_{OW} data for 50 compounds, which are listed in Table II in exactly the order reported in their paper. The entire data set are plotted in Figure 1. An obvious correlation exists as indicated by Veith's regression. Also shown is Neely's regression, which has a lower slope. The simplest test of the hypothesis that the slope n is unity is to examine values of $\log K_{OW} - \log K_B$, which is $-\log A$ in eq 11. Also included are data for 13 additional compounds obtained from the literature, the references being given in these cases.

Examination of the data indicates that much of the deviation in $\log A$ arises from a few points which are examined below.

Several compounds lie so far from the set that Veith et al. did not include them. These are designated a in Table II and it is suspected that a severe error or a mistake was made in the original work. In some cases, designated b, there are reported values of K_{OW} that differ from those of Veith and that agree better with the K_B values. We believe that the $\log K_{OW}$ values in excess of 6.0 may be suspect since most are calculated or are measured by using a HPLC technique that involves extrapolation outside the range of values measured under equilibrium conditions. Direct equilibrium measurement of partition coefficients in excess of 10^6 is exceptionally difficult since the water phase concentrations are very low. Such compounds may also experience a membrane permeation resistance because of their molecular size, thus equilibrium may not be established. Such compounds are designated c. Several compounds, designated d, are surfactants or may ionize, thus their physical state in aqueous, octanol, or lipid phases is in some doubt. In some of these cases K_B is very low and thus there is little bioconcentration potential or interest.

Removing these compounds from the list gives a mean value for $\log A$ of 1.32 and an improved standard deviation of 0.25. There is no apparent trend in $\log A$ with K_{OW}. The line corresponding to this mean value is shown in Figure 1 and has the equation

$$\log K_B = \log K_{OW} - 1.32$$

or

$$K_B = 0.048K_{OW}$$

The correlation coefficient r^2 is 0.95, which compares favorably with the Veith et al. value of 0.90.

It is thus concluded that the above correlation will satisfactorily predict K_B by using only one constant with a standard error of 0.25 in $\log K_B$, corresponding to a factor of 1.8 in K_B.

It is believed that the fundamental process observed here is one in which K_{OW} and K_B are proportional, thus n is unity. The reason that previous correlations often gave n less than unity lies primarily in the tendency to over-

Table II. Values of log K_{OW} Reported by Veith et al. (*4*) with Additional Values

chemical	log K_{OW}	exptl log K_B	correlated log K_B	comments and ref
lindane	3.85	2.67	2.53	
atrazine	2.63	<0.90	1.31	
heptachlor	5.44	4.30	4.12	
2-ethylhexylphthalate	4.20	2.93	2.88	
DASC-3	1.00	<0.32	−0.32	d
DASC-4	1.00	<0.32	−0.32	d
NTS-1	1.00	0.32-1.00	−0.32	d
BSB	1.00	<0.32	−0.32	d
FWA-2-A	1.80	<0.32	0.48	d
FWA-3-A	1.48	<0.32	0.16	d
FWA-4-A	1.20	<0.32	−0.12	d
nitrobenzene	2.93	1.18	1.61	
p-nitrophenol	2.91	2.10	1.59	
naphthalene	3.59	2.63	2.27	b,*17*
chlorobenzene	3.79	2.65	2.47	
2,4,5-trichlorophenol	3.72	3.28	2.40	a
endrin	4.56	3.17	3.24	
1,1,2,2-tetrachloroethylene	2.88	1.59	1.56	
hexachlorobenzene	5.23	3.89	3.91	
p-biphenylyl phenyl ether	5.55	2.74	4.23	a
diphenyl ether	4.21	2.29	2.89	
carbon tetrachloride	2.64	1.24	1.32	
p-dichlorobenzene	3.53	2.33	2.21	b,*15*
biphenyl	3.88	2.64	2.56	b,*10*
chloropyrifos	4.82	2.67	3.50	a
2,5,6-trichloropyridinol	1.35	0.49	0.03	
fluorene	4.38	3.11	3.06	b
dibenzofuran	4.12	3.13	2.80	a
2-chlorophenanthrene	5.16	3.63	3.84	a
phenanthrene	4.46	3.42	3.14	
2-methylphenanthrene	4.86	3.48	3.54	
heptachlor epoxide	5.40	4.16	4.08	
p,p'-DDE	5.69	4.71	4.37	
pentachlorophenol	5.01	2.89	3.69	a
hexabromobiphenyl	6.39	4.26	5.07	c
methoxychlor	4.30	3.92	3.76	b,*11*
mirex	6.89	4.26	4.34	b,*18*
hexabromocyclododecane	5.81	4.26	4.49	
hexachlorocyclopentadiene	5.51	1.47	4.19	a
heptachloronorbornene	5.28	4.05	3.96	
hexachloronorbornadiene	5.28	3.81	3.96	
Aroclor 1016	5.88	4.63	4.56	
Aroclor 1248	6.11	4.85	4.79	c
Aroclor 1254	6.47	5.00	5.15	c
Aroclor 1260	6.91	5.29	5.59	c
chlordane	6.00	4.58	4.68	
octachlorostyrene	6.29	4.52	4.97	c
p,p'-DDT	5.75	4.47	4.43	
o,p'-DDT	5.75	4.57	4.43	
1,2,4-trichlorobenzene	4.23	3.32	2.91	
5-bromoindole	2.97	1.15	1.65	
2,4,6-tribromoanisole	4.48	2.94	3.16	
N-phenyl-2-naphthylamine	4.38	2.17	3.06	a
tris(2,3-dibromopropyl) phosphate	4.98	0.44	3.66	a
tricresyl phosphate	3.42	2.22	2.10	
chlorinated ecosane	7.05	1.69	5.73	a
diphenylamine	3.42	1.48	2.10	
toluenediamine	3.16	1.96	1.84	
chloroform	1.95	0.78	0.63	*19,13*
acenaphthene	3.92	2.59	2.60	*19,16*
benz[*a*]anthracene	5.61	4.00	4.29	*19,20*
1,2,3,5-trichlorobenzene	4.46	3.26	3.14	*19,16*
trifluralin	5.34	3.76	4.02	*19,8*
pyrene	4.88	3.43	3.56	*19*
9-methylanthracene	5.07	3.66	3.75	*19,8*
benzene	2.11	1.10	0.79	*19,17*
anthracene	4.34	2.96	3.02	*19,8*
4-chlorodiphenyl oxide	4.08	2.87	2.76	*8*
4-chlorobiphenyl	4.26	2.77	2.94	*8,21*
pentachlorobenzene	5.19	3.70	3.87	*8*
dieldrin	5.48	4.11	4.16	*19,6*

estimate K_{OW} for [illegible] molecular weight compounds. With the available experimental data it is impossible to determine whether or not the slope of the $K_B - K_{OW}$ line does tend to a lower value at high K_{OW} values. Clearly there is a need for accurate K_{OW} and K_B data in this region, which can be used to test the linearity hypothesis proposed here.

It is possible that other factors such as membrane permeability will limit bioconcentration and that chemicals that bind to protein may be underestimated in analysis of biotic samples. The correlation must thus be used with care in such situations and should not be extrapolated or used for compounds that differ in chemical nature from those used in its development.

Aqueous Solubility

In some recent work Mackay et al. (*13*) analyzed the relationship between K_{OW} and aqueous solubility and developed the following equations:

$$\ln K_{OW} = 7.494 - \ln C \quad \text{for liquids}$$

$$\ln K_{OW} = 7.494 - \ln C + 6.79(1 - T_M/T) \quad \text{for solids}$$

In these correlations the solubility C has units of mol/m^3, T_M is the melting point (K), and T is the system temperature, usually 298 K. The differing treatment of solids and liquids arises from the fact that solids have lower solubilities than their subcooled liquids, the ratio of solubilities being the fugacity ratio or the ratio of the solid and subcooled liquid vapor pressures. From Yalkowsky's (*14*) estimate of the entropy of fusion this ratio can be expressed as $\exp(-6.79(1 - T_M/T))$ or $10^{-2.95(1-T_M/T)}$.

It follows that solubility should also be a satisfactory correlating quantity for K_B, as has been noted by Kenaga and Goring (*8*) and Chiou et al. (*11*). In such correlations, the solubility that should be used is C_L mol/m^3, which is either the actual solubility if the solute is liquid, or, if the solute is a solid, an estimated subcooled liquid solubility obtained from the actual solid solubility C_S as

$$C_L = C_S \exp(-6.79(1 - T_M/T))$$

Writing the $K_{OW} - C_L$ correlation in base 10 logarithms gives

$$\log K_{OW} = 3.25 - \log C_L$$

or $K_{OW} = 1795/C_L$.

In summary, the bioconcentration factor K_B can be given as either $K_B = 0.048K_{OW}$ or $K_B = 86/C_L$, where C_L is the liquid solute solubility or the "corrected" solid solute solubility as described above. An interpretation of this is that the "solubility" of a solute in a fish is 86 mol/m^3 while the "solubility" in octanol is 1795 mol/m^3.

Specimen Calculation

Carbon tetrachloride (a liquid) has an aqueous solubility of 800 g/m^3 or 5.2 mol/m^3 and a log K_{OW} of 2.64, i.e., K_{OW} is 437. K_B estimated from solubility is 86/5.2 or 16.5 and from K_{OW} is 0.048 × 437 or 20.9. The experimental K_B is 17.4, i.e., log K_B is 1.24 (*4*).

Pyrene, a solid of melting point 156 °C (429 K), has an aqueous solubility of 0.135 g/m^3 or 0.000 667 mol/m^3 and a log K_{OW} of 4.88, i.e., K_{OW} is 75 900. The fugacity ratio term at 25 °C (298 K) is $\exp(-6.79(1 - 429/298))$ or 19.8, i.e., the "corrected" liquid solubility is 0.0132 mol/m^3. K_B estimated from this solubility is 86/0.0132 or 6515 and from K_{OW} is 0.048 × 75 900 or 3643. The experimental K_B is 2691, i.e., log K_B is 3.43 (*19*).

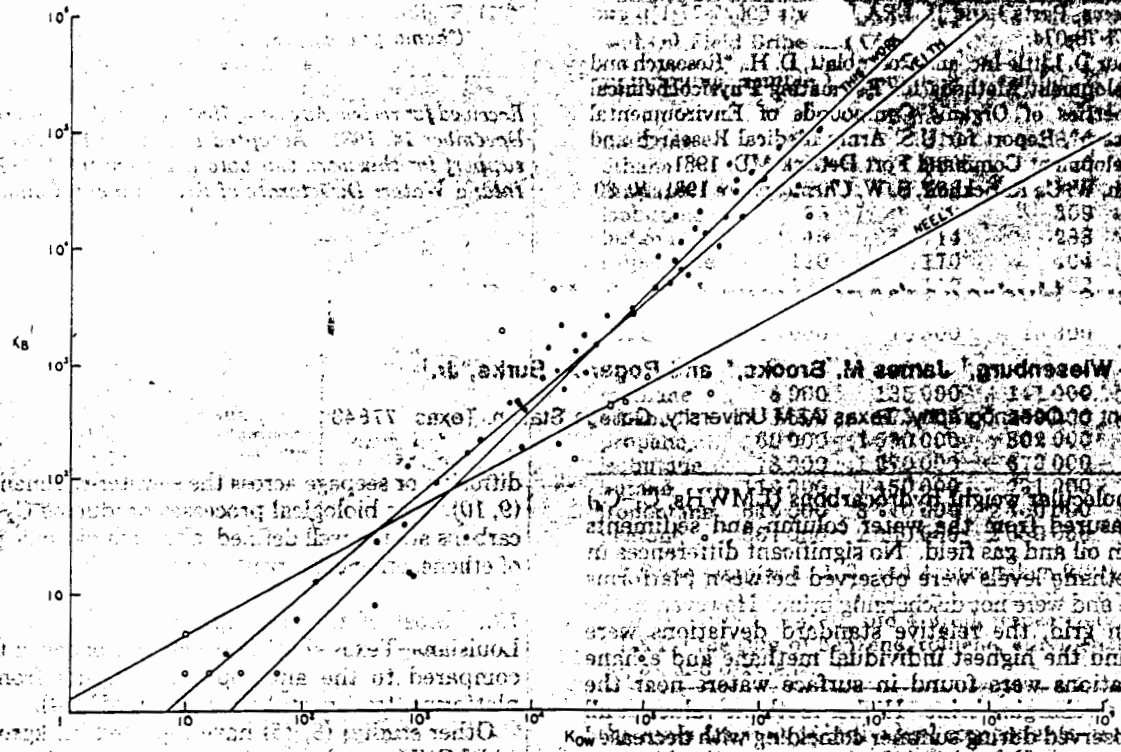

Figure 1. Plot of K_B vs. K_{OW} showing the correlations of Veith et al., Neely, and this work. Only solid points are included in the correlation.

Conclusions

Analysis of the bioconcentration phenomenon from a physical–chemical viewpoint suggests that K_B can be regarded simply as a partition coefficient between an organism consisting of a multiphase system and water. If the dominant concentrating phase is a lipid that has similar solute interaction characteristics to octanol, a proportional relationship between K_B and K_{OW} is expected. Examination of available data suggests that this is the case, especially if some suspect data are disregarded. A one-constant correlation between K_B and K_{OW} is thus suggested. This can be extended to give a simple K_B–aqueous solubility correlation in which the actual solubility is used for liquid solutes and a "corrected" solubility used for solid solutes. Although there are inherent dangers in disregarding data that do not fit a proposed correlation, it is believed that it is justified in this case, particularly as a satisfactory fit is obtained with one constant instead of the usual two. The correlation must be used with discretion. It is possible that it will fail for high-K_{OW} compounds, i.e., 10^6 and above, and the K_{OW}–solubility equation applies only to hydrophobic organic solutes of the type used to develop the correlation. For compounds of very low K_B, the amount of solute in nonlipid phases may be appreciable, and thus it is suggested that values of K_B below 10 be treated separately. Compounds that react with half-lives comparable to or less than the uptake time may be in a steady state between uptake, depuration, and degradation, and thus this purely equilibrium analysis will fail. Finally, the value of K_B is sensitive to the volume fraction and composition of the biotic phases, thus differences are to be expected between organisms and between the same organism at different life stages or conditions. A fruitful area of investigation would be to interpret K_B data for different organisms in the light of their contents of lipid and other phases and build up a more general equation for bioconcentration applicable to other organisms.

Finally, the primary contribution of this work is regarded as being a suggestion of a general alternative structure for K_B, K_{OW}, solubility correlations, and approximate values for the parameters. There is a need for more data to test these hypotheses and establish more accurate values of the parameters, especially for high K_{OW} compounds.

Acknowledgments

I am indebted to Donald Chan for help with the calculations.

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Received for review August 4, 1981. Revised manuscript received December 14, 1981. Accepted December 14, 1981. Financial support for this work was obtained from the USEPA and the Inland Waters Directorate of Environment, Canada.

Gaseous Hydrocarbons around an Active Offshore Gas and Oil Field

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■ Low molecular weight hydrocarbons (LMWHs, C_1–C_4) were measured from the water column and sediments around an oil and gas field. No significant differences in mean methane levels were observed between platforms that were and were not discharging brine. However, in the 20-station grid, the relative standard deviations were greater and the highest individual methane and ethane concentrations were found in surface waters near the platform discharging brine. Higher methane values at all depths observed during summer coinciding with decreased ethane/ethene ratios in a near-bottom nepheloid layer provided direct evidence of in situ biological production associated with increases in zooplankton and bacterial biomass in the water column. The sediment LMWHs are predominantly of thermogenic origin probably due to seepage from the subsurface, as evidenced by high levels of methane and elevated ethane/ethene ratios. The LMWH input from brine discharge in the field is estimated at 283 g/day.

Introduction

As the energy requirements of an expanding society continue to grow, there has been increased offshore drilling for oil and gas to meet those energy demands. With this increasing offshore activity, much recent concern has been focused upon the potential contamination of the marine environment by offshore production operations. An effective method for evaluating petroleum-derived hydrocarbon inputs into the marine environment is by measurement of gaseous, low molecular weight hydrocabrons (LMWHs). While LMWHs are not themselves important environmentally, because of their low toxicities, they are of secondary importance since they are introduced along with the light aromatic compounds, the most immediately toxic components of petroleum (*1, 2*). Several studies (*3, 4*) have shown that LMWHs are excellent tracers of these most toxic components of petroleum. LMWHs are also of significance in geochemical petroleum prospecting, because of the upward flux of gaseous hydrocarbons from subsurface gas and oil reservoirs (*5*).

The natural processes controlling the geochemistry of LMWHs in the water column are largely unknown, except in anoxic areas where obligate anaerobic bacteria produce methane (*6*). Concentrations of methane higher than atmospheric equilibrium values exist naturally in the aerobic water column because of in situ generation (*7, 8*) and diffusion or seepage across the sewater-sediment interface (*9, 10*). The biological processes producing C_2–C_4 hydrocarbons are not well defined, although biogenic production of ethene, ethane, propene, or propane have been observed in the water column (*11, 12*) and surficial sediments (*10, 13*). Most of the natural sources of LMWHs to the Louisiana–Texas shelf, however, are believed to be small compared to the anthropogenic inputs from offshore platforms, transportation, and runoff (*14*).

Other studies (*8, 15*) have reported background levels of LMWHs in the marine environment, as well as the resultant increases in these levels associated with isolated, catastrophic inputs that have resulted from offshore drilling accidents (*4, 16*). In this paper, we report on measurements of LMWHs around an active, established, gas- and oil-producing field in the northwestern Gulf of Mexico. The effects of anthropogenic inputs upon natural concentration levels are detailed.

The Buccaneer Gas and Oil Field (BGOF) is located ca. 50 km south southeast of Galveston, TX (Figure 1) in ca. 20 m of water. The field consists of 17 structures, of which 2 are production platforms, 2 are quarters platforms, and 13 are satellite structures surrounding well jackets. Produced brine from the production operations is discharged into the surface waters at only one of the production platforms (platform 296B). The brine discharge rate has been estimated at 95 400 L/day (*17*). The field has been in production since 1960. A unique feature of BGOF is that it is relatively isolated from other production fields, thus our study was not complicated by other nearby anthropogenic hydrocarbon sources.

Experimental Section

Samples of the produced brine and oil were collected for analysis of LMWHs along with seawater and sediment samples from a grid of stations around platforms 288A and 296B (see Figure 1). Water column samples were taken from the surface and from 10- and 20-m depths around both production platforms. The 20-station grid for water sampling comprised four lies extending north, south, east, and west of each platform, with samples being collected at 25, 50, 100, 150, and 300 m distant from the platform. Sediment samples were taken along the same grid lines at distances of 10, 25, and 50 m.

Water column sampling was performed aboard the M/V Tonya and Joe by using standard 12-L Niskin sampling bottles. Samples for LMWHs were transferred into 200-mL glass bottles, which were allowed to overflow, poisoned with sodium azide, and capped without a headspace. The samples were analyzed in a shore-based laboratory after the methods of Brooks et al. (*8*). The method involves stripping the hydrocarbons from solution with a purified

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0013-936X/82/0916-0278$01.25/0 © 1982 American Chemical Society