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RECREATIONAL RESOURCES AND WATER QUALITY

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INTRODUCTION

Recreational water quality is an issue of current interest in Canada and the United Kingdom. Both nations have standards for recreational waters based upon extensive epidemiological research which has 'defined' the link between bathing in sewage polluted water and disease occurence in the recreator population (E.E.C., 1976; Canadian Government, 1983). The implementation of policies based upon these standards has caused considerable debate on both sides of the Atlantic because of the economic impact on the recreation industry that beach closure decisions can produce. In addition, the imposition of water quality objectives on the sewage disposal industry has associated cost implications which have to be considered in any assessment of overall impact. The responses of Canadian and UK managers to these cost implications have been very different. In Canada, the stated policy is applied to recreational waters and beaches are closed regardless of the political and economic costs involved (Horgan, 1983; Hollobon, 1983). In Britain, a series of mechanisms have been devised which prevent the implementation of water quality standards to the vaste majority of bathing beaches (McDonald and Kay, 1984). Proponents of the opposing Canadian and British policies cite conflicting epidemiological evidence to support their cases. However, much of this debate rests on simplistic and incorrect assumptions concerning the epidemiological risk implied by a given concentration of the water quality parameters under test. The major parameters used are the faecal indicator bacteria (faecal coliforms, _Escherichia coli_ and enterococci). These are non-conservative pollutants which demonstrate extreme temporal and spatial variability determined by input rates and a complex mix of environmental paramters which combine to produce bacterial decay, regrowth or prolonged survival.

The ability to model and accurately predict these changes in water quality is central to effective policy implementation but such an ability does not at present exist. This paper contrasts the current U.K. and Canadian policy and evaluates potential modelling strategies used to predict pollution levels in the nearshore bathing zone.

EDIDEMIOLOGICAL EVIDENCE

Two major investigations provide the scientific evidence on which estimates of 'risk' to bathers from sewage polluted recreational waters are based.

United Kingdom Studies

The first of these studies was a retrospective epidemiological investigation into the incidence of enteric fever and poliomyelitis conducted by the British Public Health Laboratory

Service (P.H.L.S., 1959). This study examined a narrow range of two notifiable diseases and failed to establish a statistically significant relationship between disease incidence and bathing. Subsequent interpretations of the P.H.L.S. survey have suggested that the absence of any proven link between the diseases studied and bathing events indicates that little or no risk is attributable to bathing in sewage polluted waters (Medical Research Council, 1959). This erroneous interpretation has been restated by Giles Shaw in a written Parliamentary answer given in his capacity as environment minister in 1981 (Pearce, 1982; McDonald and Kay, 1984). Whilst this view is strictly speaking true, the lack of positive evidence does not prove a lack of **risk** since risk derives from a much broader range of diseases than the two notifiable illnesses studied.

North American Studies

North American investigators in Canada and the United States have attempted to define this level of risk employing a superior experimental design to the earlier U.K. research. These studies have employed a prospective cohort design in which bathers at risk of ingesting polluted water by immersion of their nasal or oral orifices are identified at the time of bathing (Cabelli et al., 1982; Dufour, 1982,1983). Similar cohorts of non-bathers are also identified and then disease incidence in both groups is monitored over a two week period by telephone and questionnaire survey. The advantage of this methodology over the earlier British research is that all disease occurance can be monitored and water quality determinations can be completed to define the functional relatioship between water quality and bather morbidity.

The problems of the prospective cohort design are that very large samples are required if statistically significant differences are to be identified between the bathing and non-bathing cohorts. The work of Cabelli et al. (1982) and Seyfried (1980) illustrate how the prospective cohort design is applied in the U.S.A and Canada respectively. Significant regression relationships have been defined by Cabelli et al. (1982) which suggest that indicator densities of 2000 FC $100ml^{-1}$ can result in additional disease incidence at rates of approximately 30-40 cases displaying gastroenteritis symptoms per 1000 bathers. This indicator density is in fact the upper 'IMPERATIVE' level set by the E.E.C. under the 1976 Bathing Waters Directive.

The Scientific Debate

To many British scientists it seemed that Cabelli's results were in direct contradiction to the earlier P.H.L.S. investigation. This has led to several statements by the competent U.K. authorities which have either questioned the scientific validity of Cabelli's work or dismissed the gastroenteritis infections identified as too minor to merit serious public health concern. This latter position was taken by Barrow who stated;

> "It is clear that the risk of contracting serious disease is minimal, whether or not bathers are subject to more 'ailments' than non-bathers."
>
> (Barrow, 1981:223)

and by the Welsh Office who have recently stated;

> "There is, however, no evidence that sewage pollution (as indicated by high coliform counts) in the waters off Welsh

beaches results in the spread of serious illness."
(H.M.S.O., 1985:9)

The former argument, attacking the statistical validity of the North American research, dates from an authoritative review by Moore (1975) who served as the chairman of the P.H.L.S. committee. Subsequently the Welsh Water Authority have utilised the statistical argument in their written evidence to the parliamentary enquiry cited above. Their submission rests heavily on a review paper by Stanfield of the Water Research Centre (Stanfield, 1982) who is cited as evidence that the P.H.L.S. study is still the most relevant work in this area. This was in fact a strange interpretation of Stanfield's paper which stated;

> "Certainly the results of the P.H.L.S. survey no longer appear to provide the assurance demanded by the general public and by organisations with direct responsibilities for coastal waters."
>
> (Stanfield, 1982:53)

Further evidence of dissatisfaction with the P.H.L.S. study comes from the Royal Commission on Environmental Pollution, which stated in its tenth annual report

> "it is now necessary to modify the reliance placed on a report published almost a quarter of a century ago."
>
> (H.M.S.O., 1984:87)

The current position of the competent British Authorities is one of some confusion and disarray. The complimentary epidemiological investigations of the P.H.L.S. and Cabelli et al. are often incorrectly assumed to be contradictory and hence requiring defence or support rather than evaluation and assessment.

THE 'STANDARDS' DILEMA

Recreational access policy requires agreement on water quality standards which are applicable to many locations and adequately define the epidemiological risk to bathers. Herein lies the second major area of controversy. The E.E.C. specify some nineteen parameters of recreational water quality including inert floating materials and dissolved toxins such as cyanide. However, the common parameter in all monitoring programmes are the faecal indicator organisms and in particular the coliform group. A more epidemiologically significant subset of this group, the faecal coliforms, are now used in regular water analysis to indicate the risk of pathogen presence. Underpinning all such 'risk' estimates are assumptions regarding the likely indicator pathogen relationships experienced in the environment. In effect, such consistency does not occur because indicator bacteria may reach the bathing zone from many different sources which all have unique indicator pathogen relationships. Potential sources can include;

(i) sanitary sewer outfalls,

(ii) combined stormwater systems,

(iii) natural stream channels,

(iv) birds and

(v) bathers.

Each bathing location will experience a different and constantly changing mix of these inputs and will thus experience changing indicator pathogen relationships even over very short timescales of a few hours. This throws considerable doubt on the universal application of indicator densities as surrogates for risk in the bathing zone.

The second reason for a relatively circumspect attitude to the use of bacterial indicator species for defining risk to recreators is the great spatial and temporal variations in the indicator densities themselves. Spatial variablity is caused by point sources (i to iii above) which produce elevated concentrations in close proximity to the input. There is a further consideration in this complex situation because the type of point source partly determines the nature and degree of variablity experienced. Streamwaters produce elevated bacterial concentrations during high flow events (Qureshi and Dutka, 1979; McDonald and Kay, 1981; McDonald et al., 1982). Similar elevated concentrations are experienced in urban runoff from combined sewerage systems. This rainfall generated input contrasts with the typical flow from a sanitary sewer which often discharge for regular fixed time periods determined by holding tank capacities and operative work schedules.

Data illustrating this variability at Aberaeron on the West Wales Coast (U.K.) and Lake Ontario beaches are presented in table 1. Observations of this type of data variability have led some researchers to comment that it is possible to artificially produce an apparent upgrading or downgrading of a beach as a recreational resource by judicious sampling programme design (Barrow, 1981).

THE POLICY RESPONSE

The response of authorities to this problem has been to design standards which attempt to characterise water quality over an extended time period. For example the Toronto Public Health Department determine the ten day geometric running mean faecal coliform count for a series of water samples. If this value exceeds 100 $.100ml^{-1}$ then the beach should be closed due to poor water quality. The Canadian federal standards suggest that the geometric mean value of 5 samples taken over a 30 day period should be less than 200 faecal coliforms $.100ml^{-1}$ and that immediate resampling should occur if any one sample exceeds 400 faecal coliforms $.100ml^{-1}$. Six different standards operate within Canada all of which are more stringent than the current European standards defined in the E.E.C. Bathing Waters Directive (E.E.E., 1976). The E.E.C. document sets out '**Guide**' and '**Imperative**' values for recreational waters and the faecal coliform standards are set out in table 2.

Neither the North American or European standards have any epidemiological rationale. They can not be directly related to either the P.H.L.S. or indeed Cabelli's research because 'risk' for an individual bather is related to water quality at the time of ingestion. It is possible for a beach to have extremely high 'spot' concentrations of faecal indicators but for the beach still to be classified as satisfactory on the basis of the above criteria. These short term high concentrations could either be missed due to poor sampling programme design or their statistical effects on the measures of central tendency used (geometric mean or median values) masked by a run of much cleaner samples.

MODELLING STRATEGIES

The scientific problems involved in setting water quality standards have led many authorities to question their implementation and caused considerable controversy when beach closures have resulted from the standards outlined above (Barrow, 1981; Moore, 1975; Hollobon, 1983). It is all the more difficult to explain the rationale for closure to economically injured parties when the standards used relate to past water quality and may not therefore reflect the actual risk on the day of closure. An exact definition of this risk, even in terms of indicator concentrations, is not possible because the minimum time required for a faecal coliform enumeration is 18 hours, far longer than the typical bathing day (H.M.S.O., 1983; Canadian Government, 1983).

Predictive coliform modelling presents one avenue by which times of high and low risk can be identified. Two main model types can be identified namely process simulation models and statistical models. The former have been used to predict bacterial concentrations in small streams by Jenkins et al. (1984) and to predict beach pollution levels in the Ottawa River by Palmer and Dewey (1984). This method provides a sound base for sewage outfall design and provided resources are available for a hydrographic survey simulation modelling can produce satisfactory estimates of beach water quality. However, this method does not respond well to short term episodic changes in the bacterial loadings caused by rainfall events. Associated with hydrograph events and the consequent ´flush´ input of indicator organisms rapid changes occur in the receiving environment which themselves have a significant effect on the bactericidal nature of the bathing waters. Simulation models quantify the bactericidal nature of the receiving environment by assuming a constant rate of decay (a T_{90} value of 17 hours in the case of Palmer and Dewey, 1984). The bactericidal process is extremely complex and is not sufficiently understood at present to allow the exact effects of say increased turbidity and decreased light penetration to be modelled. Current process models do not therefore provide a good predictive capability of the short term episodic changes in beach enteric bacterial concentrations caused by rainfall events.

The statistical models attempt to incorporate these environmental parameters using a multiple regression approach. This method has been used successfully to predict impoundment coliform concentrations (Kay and McDonald, 1983) and it offers the potential for predicting enteric indicator concentrations at bathing sites using only a few easily measured environmental parameters. Table 3 presents data on 35 regression equations calculated for a number of bathing locations on the shores of Lake Ontario in the Metropoliton Toronto area. The predictor variables chosen for inclusion are all based on meteorological measurements. R1 to R11 represent rainfall totals on the sampling day (R1), the previous day (R2) and so on. S1 to S11 represent hours of sunshine for corresponding time intervals. MAXWIND and AVEWIND are the maximum and average wind speeds on the day of sampling. This list of independent variables represents the total universe of data from which a predictor variable subset could be chosen. The three broad categories of predictor, namely; rainfall, sunshine and wind magnitude, were chosen because of the established relationships between rainfall and peak enteric bacterial concentrations (Toms et al., 1981; McDonald and Kay, 1981); the known bactericidal effects of sunlight (Gameson and Saxon, 1967) and the hypothesised wind induced disturbance of the sediment associated bacterial store (Babinchak et al., 1977; Jenkins and McDonald, 1985).

Control of the predictor variable subset used in each equation was effected using stepwise inclusion under the constraints defined by the parameters outlined in Nie et al. (1970). A maximum of ten predictor variables was allowed in each equation and a tolerance value of 0.5 was chosen to limit the degree of allowable multicollinearity. The minimum F level allowed was 2.0 (where $F=(B/\text{Standard Error of } B)^2$. Skew in the stochastic disturbance term was reduced by a $\log_{10}$ transformation of the dependent variable (Poole and O´Farrell, 1971; Velz, 1951; Gameson, 1981).

The coefficients of determination (R^2 terms) show a range from 0.19 to 0.81 with a mean value of 0.49. If we examine the predictive value of equations for discrete groups of these sampling locations a distinct pattern emerges. Three main areas can be identified. The first stretches from the Humber River in the west to Ontario Place, the second comprises a number of sampling locations around the Toronto Islands and the third contains some sixteen sampling stations all on beaches to the east of Coatsworth Cut starting at Woodbine Beach and ending at Willow Avenue in the east. The western group are characterised by high R^2 values with a mean R^2 of 0.67 and a mean standard error of 0.50. The twelve Toronto Island sampling sites have a mean R^2 of 0.58 and a mean standard error of 0.50. The regression equations describing the eastern beaches have lower predictive capability than either of the previous groups. Here the mean R^2 drops to only 0.35 and the standard error term increases to 0.68. All equations, however, are statistically significant. Based upon the equation F levels, only one equation demonstrates lower than 99% significance and 25 of the 35 equations show a significance level of 99.9%.

DISCUSSION

Stochastic variance, not accounted for by the prediction equations listed above, may derive from many sources. Avian inputs are a well accepted sources of both pathogens and indicators (Palmer, 1983; White, 1985). In the Toronto area, additional unexplained inputs of bacterial indicators will be evident from the sanitary sewer outfalls. Here flow regimes and hence bacterial inputs may not be related to the hydrometeorological predictor variables used in the equations. These factors go some way to explaining the discrepancy between the western and eastern groups of sampling stations. In the east, no major riverine input affects water quality and there are six sanitary sewer inputs to the bathing zone over a 1.2 km stretch of shoreline. In the western area only two sanitary sewers discharge into the bathing zone over a beach length of some 2.6 km and the Humber River provides a major input of faecal indicators. This input is extremely variable and will be determined by the flow regime of the river which should be partly explained by the meteorological predictor variables included in the regression equations generated. The lower predictive capability of the regression equations describing the eastern locations is an important flaw in the models presented because the sanitary sewer outfalls are likely to produce pollution with higher levels of human pathogens and hence similar indicator concentrations might not indicate similar levels of risk. The avian inputs to the bathing zone will remain very difficult to measure. However, future modelling strategies should aim to incorporate parameters describing the discharge rates of sanitary sewers

The predictive capability of the equations presented in this paper are broadly in line with previous studies using multivariate methods to predict enteric indicator species (Kay and McDonald, 1983; Jensen et al., 1981). A direct comparison of statistical model with the simulation approach adopted by Palmer and Dewey (1984) is not possible because only five data points defining predicted and actual beach water quality are presented by these authors (Palmer and Dewey, table 1).

CONCLUSIONS

In both Canada and the United Kingdom recreational water quality is an issue of growing concern because of the identified link between water quality and the incidence of gastrointestinal infections. This concern is likely to lead to increasing pressure to restrict recreational activity at polluted locations. The difficulty facing the managers of these rural recreation sites is that present monitoring programmes are not capable of providing information on the levels of 'risk' experienced by bathers actually at the time of exposure. Hence, decisions to close a beach because of past water quality may seem perverse and meaningless to the economically disadvantaged recreational industry. In the absence of instantaneous bacterial enumerations, estimates of risk must be based on predictive modelling to define the future periods of poor water quality. The multivariate approach adopted here provides good predictive capability for one set of beaches affected by riverine inputs. Further work is required to explain the water quality in the bathing zone at locations dominantly affected by sanitary sewer discharges. Although the application of more transferable simulation models to recreational waters is a desirable long term objective the multivariate approach offers a useful predictive tool which can be applied with little initial cost.

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TABLE 1
Spatial variability of bacterial counts .100ml^{-1} at two bathing beaches in Ontario and Wales UK. Source for Lake Ontario data Totonto Department of Public Health.

	SITE	FAECAL COLIFORM 100 ml^{-1}
Lake Ontario 23.5.84	1. Humber River	25,000
	2. Windermere	30,000
	3. Ellis Ave	21,000
beach length 2.6 km	4. Sunnyside	960
	5. Boulevard Club	26
	6. Argo Rowing Club	22
Aberaeron beach 20.8.83	1. River Aeron	44,000
	2. North beach A	8,100
	3. North beach B	8,600
	4. North beach C	6,100
	5. North beach D	4,500
	6. North beach E	1,000
beach length 1.5 km	7. North beach F	1,830
	8. North beach G	1,000
	9. North beach H	1,200
	10. South beach	90

Aberaeron samples 2-9 taken at approximately 50m intervals commencing at the Aeron mouth in the south and sampling at the groynes running transverse to the beach.

TABLE 2
Current E.E.C. bathing water quality standards in terms of coliform bacterial concentrations. Source E.E.C. (1976).

PARAMETER	**GUIDE**	**IMPERATIVE**
total coliform	500	10,000
faecal coliform	100	2,000

All figures in bacteria $.100ml^{-1}$. Samples should be collected fortnightly and 95% of the samples should be less than the **IMPERATIVE** value. 80% of samples should comply with the **GUIDE** value.

TABLE 3
Regression equation statistics and predictor variables chosen by the stepwise inclusion criteria specified in the text. SE is the standard error of the stochastic disturbance term and R^2 is the coefficient of determination

SAMPLING LOCATION	SITE NO.	R^2	% SIG	SE	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	R11	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	AVEWIND	MAXWIND
WESTERN BEACHES	9401	.79	0.1	.35							*	*				*		*					*	*	*			*
	9402	.48	1.0	.65		*										*		*							*			
	9405	.59	1.0	.60		*		*						*				*										*
	9406	.66	0.1	.64							*	*			*	*	*						*	*				
	9407	.73	0.1	.46		*			*					*		*	*	*								*		
	9410	.79	0.1	.35							*	*				*		*					*	*	*			*
TORONTO ISLANDS BEACHES	9415	.55	0.1	.58													*	*					*			*		
	9430	.78	0.1	.36		*	*					*		*	*	*		*						*	*	*		
	9431	.34	2.5	.57		*										*		*										
	9435	.40	1.0	.56	*										*			*										
	9440	.59	0.1	.46	*		*		*									*	*									
	9441	.32	1.0	.77	*													*										
	9445	.72	0.1	.43	*				*			*		*		*		*			*					*		
	9450	.81	0.1	.35				*		*	*	*								*	*			*		*	*	
	9460	.74	1.0	.35							*			*	*	*	*	*			*	*			*	*		
	9470	.68	1.0	.36				*							*		*		*			*		*			*	
	9471	.61	1.0	.57			*	*	*									*				*				*	*	
	9480	.49	1.0	.58													*			*		*				*	*	
EASTERN BEACHES	9501	.36	0.1	.58		*					*			*		*									*	*		
	9502	.38	0.1	.59			*					*				*	*			*				*		*		*
	9503	.29	0.1	.66	*								*			*				*	*			*				*
	9504	.48	0.1	.51			*					*		*		*	*			*				*	*			*
	9505	.47	0.1	.53		*						*		*		*	*			*	*			*				
	9506	.44	0.1	.58		*					*	*			*	*	*							*				
	9507	.42	0.1	.72	*	*								*		*	*								*	*		
	9508	.35	0.1	.71		*	*							*		*							*				*	
	9509	.33	0.1	.70	*	*		*								*											*	
	9510	.19	1.0	.74		*		*			*					*	*											
	9511	.48	0.1	.62		*	*	*		*	*	*					*				*			*		*		
	9512	.25	0.1	.84								*				*								*				
	9513	.30	0.1	.73		*										*					*						*	
	9514	.37	0.1	.66		*	*			*						*					*						*	
	9515	.36	0.1	.69	*			*						*		*		*	*		*	*		*				*
	9516	.29	0.1	.76		*								*						*							*	
	9517	.34	0.1	.68	*	*										*	*											*