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MODELLING THE EFFECT OF DROUGHT ON ESTUARINE WATER QUALITY

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Abstract—Long-term data assembled for monitoring the water quality of the Thames estuary between 1977–1992 were used to examine the changing influences of physico-chemical variables on estuarine water quality during drought conditions. Seasonal and monthly means were computed and tested for significant differences. Time series data were used to estimate regression models explaining observed variations in water quality parameter measurements during the predrought (1977–1988) and drought (1989–1992) periods. Stability tests were used to establish significant differences in predrought and drought models and to make inferences about the significance of drought induced low flows on water quality. Significant mean seasonal and monthly differences were found for flow, chlorinity, temperature, dissolved oxygen, pH and suspended solids. Total oxidisable nitrogen showed no significant change. Regression analyses were consistent with these results and highlighted the significance of drought related changes in flow and temperature for the determination of estuarine water quality. This suggests that management policies designed to mitigate the effects of drought be extended to consider their effects on estuarine water quality given the ecological importance of these areas. © 2000 Elsevier Science Ltd. All rights reserved

Key words—Thames, flow, temperature, pH, chlorinity, solids, oxygen

INTRODUCTION

Periods of drought can have dramatic effects on aquatic systems by reducing the quantity of freshwater within lakes (Webster *et al*., 1996) and water level and flows in rivers (e.g. Tallaksen *et al*., 1997). This can be compounded by the requirements of humans for drinking water (and for other activities), with additional water resources being removed from the water body (Harding *et al*., 1995). The ultimate consequence of these actions can be a reduction in the level of ground water and the complete drying-up of rivers (Miles, 1993). As well as impacting the quantity of water within rivers, reductions in flow can also affect water quality due to diminished levels of flushing within the system (De Mars and Garritsen, 1997). This impact on river water quality has received some attention (e.g. Chessman and Robinson, 1987; Welsh and Stewart, 1989; Barros *et al*., 1995) particularly where additional abstraction is practiced (e.g. Muchmore and Dziegielewski, 1983).

Estuaries by their very nature require a significant input of freshwater from rivers (Pritchard, 1967), and cyclical patterns of salinity and other physico-chemical parameters are dependent on seasonal flow characteristics (Thatcher, 1992; Attrill and Thomas, 1996; Kurup *et al*., 1998). Additionally, water quality within urbanized estuaries can be potentially influenced by changes in flow by increasing or decreasing the residence time (and hence flushing) of pollutants within an estuary. Consequently water quality measurements, such as dissolved oxygen, tend to be lower in summer due to a combination of low flows (preventing flushing of organic waste) and increased temperature (accelerating the rate of bacterial breakdown) (Breitburg *et al*., 1997; Kinniburgh, 1998). A relatively small number of studies have investigated the effect of seasonal (Chapman and Brinkhurst, 1981; Wells and Young, 1992) and spatial (Montagna and Kalke, 1992) variations in freshwater flow on estuarine ecosystems. Even fewer studies have examined the impact of drought-induced flow on estuaries. Those that have, focused on specific components of the biota (e.g. Andrews, 1977; Jones, 1990; Bennet *et al*., 1995; Attrill *et al*., 1996). Thus, in comparison with river systems, the effect of drought-induced low flows on the water quality of estuaries has been understudied.

Due to low precipitation, drought conditions per-

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Table 1. List of measured physico-chemical variables, together with reference for the relevant analytical method where appropriate

Parameter	Reference
Freshwater flow	n/a (Gauged flow over Teddington Weir ($m^3 s^{-1}$))
Temperature	n/a (Thermometer)
Chlorinity	EPA 600/4-79-020 Method 353.3
Dissolved oxygen	HMSO (1980a)
pH	HMSO (1979)
Suspended solids	HMSO (1980b)
Total oxidised nitrogen	EPA 600/4-79-020 Method 358.2

sisted in the south-east of England for a four year period from 1989–1992 (Boar *et al*., 1995). The severity of the drought in the Thames catchment was augmented by an increase in average abstraction rates, making river flows during the period amongst the lowest on record (Littlewood and Marsh, 1996). Regulations requiring a minimum rate of 200 tcmd ("thousand cubic metres per day" $\equiv 2.31\ m^3 s^{-1}$, see Sexton, 1988) prevented flows from reaching possible no-flow conditions as occurred during the 1976 drought (Attrill *et al*., 1996). Nevertheless, freshwater flows into the Thames estuary during the drought period were much reduced and the patterns of salinity distribution further down the estuary were significantly affected (Attrill and Thomas, 1996). It is possible, therefore, that the persistence of the low-flows resulted in other drought-induced changes in water quality within the estuary.

The long history of concern over pollution and recovery of the Thames (e.g. Wheeler, 1979; Tinsley, 1998; Power *et al*., 1999a) has resulted in the assembly of a comparatively long-term data set of water quality parameters by the Environment Agency and precursory organizations. As a consequence, a comprehensive data set covering both nondrought (1977–1988) and drought (1989–1992) periods is available for use in a comparative study of drought-induced changes in estuarine water quality. The aim of this paper, therefore, is to assess the impact of drought-induced low flows on a suite of water quality parameters (temperature, chlorinity, dissolved oxygen, pH, suspended solids and total oxidised nitrogen) known to hold significant implications for the estuarine biota by comparing available data on water quality in the Thames estuary from drought and predrought periods.

SAMPLING METHODOLOGY

Surface water quality parameters for the study were obtained from the water analysis runs completed by the National Rivers Authority (now Environment Agency) sampling vessel along the length of the Thames estuary between January 10, 1977 and November 13, 1992. All measures used were taken at West Thurrock (National Grid Reference: TQ 593 770) 67 km downstream of Teddington weir (the input of freshwater from the river Thames). Parameters measured included: salinity (as chlorinity (mg/l)), temperature (°C), dissolved oxygen (% saturation), pH, suspended solids (mg/l) and total oxidisable nitrogen (mg/l), hereafter referred to as nitrogen. The measurement database was supplemented with daily flow rates ($m^3 s^{-1}$) of freshwater into the estuary from the river Thames measured by a fixed gauging station at Teddington weir. Chlorinity was mid-tide corrected to allow for valid temporal comparisons. All other measures were used as reported. Although other determinants (e.g. BOD, phosphate and chlorophyll) were measured, they were measured too infrequently to be considered for use in the temporal analysis completed here. All water quality determinants used in the study are listed in Table 1, together with the reference for the relevant analytical methodology.

Sampling was completed throughout the year at varying intervals, increasing in frequency during summer as a result of concerns about seasonal changes in water quality. The mean and modal times between samples were 2.7 and 2.0 weeks respectively, with 84% of all samples being separated by between 1.0 and 3.0 weeks. Minimum and maximum times between sample dates were 0.7 and 14.6 weeks respectively. The latter occurring in the winter of 1988 coincident with the reduction in water quality sampling frequency for the period 1986–1988. All sample dates for the period under study were converted to a time index variable (weeks) measured from 1.1.1977.

The period 1989–1992 has been treated in the literature as atypical due to the drought conditions that prevailed in the Thames catchment (Attrill *et al*. 1996). To assess the possibility that low freshwater flows resulting from lower precipitation and increased water abstraction may have had significant effects on water quality, analyses in this paper were divided into a 12 year predrought period extending from January 1977 to December 1988 and a 4 year drought period extending from January 1989 to November 1992.

MODELLING DETAILS

To assess possible differences between predrought and drought periods, predrought and drought summer (June 21–September 21) and winter (December 21–March 21) means for all water quality parameters were computed. Data used in the computations were tested for normality using the Shapiro-Wilk W statistic before selection of the statistical tests used to establish any possible significant difference between the periods. The prevalence of non-normal data necessitated the use of the Mann–Whitney U statistic to test the statistical significance of observed differences in mean water quality parameter values on a seasonal basis. The statistic is useable to test differences between random samples when normality cannot be assumed and its asymptotic relative efficiency is considered to be good (Conover, 1980). To establish the significance and monthly pattern of possible effects, predrought water quality data were used to compute representative monthly means and associated 95% confidence intervals for each of: chlorinity (mg/l), temperature (°C), dissolved oxygen (% saturation), pH, suspended solids (mg/l), total oxidisable nitrogen (mg/l) and daily flow rates ($m^3 s^{-1}$). Results were then compared to the monthly means computed for drought period water quality data.

Multiple linear regression methods were used to examine the significance of associations between measured water quality parameters and other available physico-chemical variables in the drought and predrought periods. The models postulate that water quality measurements respond to the physico-chemical conditions prevailing in the estuary at a given point in time. Accordingly, models of the form

$$Y_t = a_0 + a_1 X_{1t} + a_2 X_{2t} + \ldots + a_k X_{kt} + \epsilon_t$$

were estimated where Y_t represents available water quality data at time (t), X_1 to X_k are the set of physico-chemical variates used to explain variability in Y and ϵ is a normally distributed error term with zero mean and unit variance (Draper and Smith, 1981). For model estimation, logarithmic (natural) transformations were completed on data where necessary to ensure data conformed to an approximate normal distribution (Draper and Smith, 1981). Outliers, resulting in nonnormal residuals, were removed from model estimation data sets on the basis of box and whisker plots and Grubbs' test (Grubbs, 1969). For the predrought data set this resulted in the removal of 15, 3 and 5 data points, respectively, from the nitrogen, pH and suspended solids data sets. For the drought data set the procedure resulted in the removal of 1, 3 and 2 data points, respectively, from the same data sets.

To avoid problems associated with possible spurious correlation *a priori* rationalizations were used to select a feasible set of variables for possible inclusion in the multiple regression models. For example, oxygen solubility - temperature relationships allow temperature to influence dissolved oxygen, but not visa versa. Accordingly, temperature was considered as a feasible explanatory variable for the oxygen model, but oxygen was not considered as a feasible explanatory variable for the temperature model. From amongst the feasible set of variables for a particular model, the variables ultimately used to estimate the model were selected using forward selection step-wise regression and backward elimination techniques (Draper and Smith, 1981; Dunn and Clark, 1987). Forward selection selects from among the feasible variables, the independent variable with the highest F-value and enters it in the model provided the associated F-value for the variable exceeds a prespecified F-to-enter criterion. At successive steps, previously entered regressors are retained in the model providing their associated F-values do not subsequently fail to exceed a prespecified threshold (F-to-remove). Here the F-to-enter and F-to-remove values used were at the upper $a = 0.05$ point of the F-distribution as recommended by Draper and Smith (1981). Differing values of the F-to-enter and F-to-remove were used to test the effect of the choice of criteria on the subset of model variables selected and found to have no effect. Backward elimination was further used to assess the stability of the selected independent variable set used in each regression and found to have no effect on variable selection. Instrumental variables were used to capture seasonal influences on fluctuations in water quality parameters. Variables representing each month were set equal to one if data were collected for that month, zero otherwise (Koutsoyiannis, 1977).

The Chow test (Chow, 1960) was used to establish the stability of estimated predrought and drought models to changes in their respective estimation data sets. For predrought models the test was carried out by expanding the estimation data set to include all measurements made during the drought. Drought models were similarly tested by expanding their data estimation sets to include all measurements taken in the predrought period. The Chow test is a F-statistic based test. Test significance establishes that changes in the underlying structure of the model have occurred as a result of some event (Koutsoyiannis, 1977) and in this instance would establish the significance of drought effects on measured water quality parameters.

The statistical assumptions underpinning ordinary least squares were verified by examining model residuals for normality, serial correlation and heteroscedasticity. Royston's extension of the Shapiro–Wilk W statistic (Royston, 1982), applicable to sample sizes between 7 and 2000, was used to judge normality in model residuals. The Shapiro–Wilk statistic and its extensions have been recognized as amongst the most powerful omnibus tests for normality (D'Agostino, 1986). Normalized versions of the Royston statistic were computed (Royston, 1982) and may be compared to tabular values for the standard normal distribution to determine sig-

Table 2. Seasonal means and inter-quartile range for each of the measured water quality parameters. Significant differences between mean predrought and drought water quality measures for winter and summer established using the Mann–Whitney U statistic. The *p*-values for differences which are significant at the a = 0.05 level of significance are italic. Summer includes values for the period June 21 to September 21 period and winter includes values for the period December 21 to March 21

Variable	Winter			Summer		
	predrought mean	drought mean	*p*-value	predrought mean	drought mean	*p*-value
Flow ($m^3 s^{-1}$)	113.5 (62.0–144.5)	84.7 (34.0–134.0)	*0.036*	29.4 (14.0–39.0)	10.8 (6.2–11.0)	*0.001*
Chlorinity (mg/l)	5136 (3722–6200)	7543 (5000–9700)	*0.001*	9539 (8700–10,450)	12,350 (11,200–13,090)	*0.001*
Temperature (°C)	7.64 (7.00–9.00)	9.27 (8.00–10.00)	*0.001*	19.48 (18.50–20.00)	20.38 (20.00–21.00)	*0.013*
Dissolved oxygen (% saturation)	64.8 (59.0–71.5)	59.9 (53.0–70.0)	0.234	44.9 (39.0–50.5)	51.8 (46.3–56.8)	*0.005*
PH	7.45 (7.30–7.60)	7.53 (7.50–7.60)	0.050	7.41 (7.20–7.50)	7.70 (7.50–7.80)	*0.001*
Suspended solids (mg/l)	97.9 (47-5-127.5)	113.1 (44.3–158.8)	0.908	81.4 (52.0–97.5)	57.2 (33.0–65.0)	*0.017*
Nitrogen (mg/l)	9.4 (8.4–10.0)	8.8 (7.8–9.3)	0.243	6.7 (5.7–7.1)	6.5 (5.5–7.7)	0.801

nificance. Serial correlation was assessed using the runs test (Law and Kelton, 1991). The test is a direct assessment of the independence assumption for model residuals applicable when observations are not equally spaced in time. Values of the test statistic were computed following the procedure outlined in Banks and Carson (1984). Finally, homoscedastic residuals were verified by plotting standardized residuals against fitted model values and examining the resulting plots for evidence of increasing or decreasing variance, as recommended by Draper and Smith (1981).

RESULTS

Mann–Whitney tests demonstrated seasonal differences between drought and predrought periods in mean flow, chlorinity and temperature measures in both summer and winter (Table 2). Mean dissolved oxygen, suspended solids, pH and nitrogen measurements did not differ between the predrought and drought periods during the winter, although in the summer, mean dissolved oxygen, suspended solids and pH measures did differ significantly between the predrought and drought periods. No significant difference in nitrogen in either summer or winter was found between the predrought and drought periods. Figure 1 presents mean monthly data for predrought and drought periods, highlighting the significant differences in monthly measurements. Winter and summer declines in flow are particularly apparent during early seasonal periods, December–January and June–July respectively. Chlorinity increases in all months, whilst significant seasonal temperature changes are dominated by warmer late winter (January–March) and late summer (August–September) periods. Although dissolved oxygen decreases substantially in January in association with increased water temperatures, the seasonal effect is negligible owing to more typical values being recorded in the February and March period. Increases in dissolved oxygen in the June through September period, particularly in June and August, result in increases in the summer seasonal mean despite the decreases that would be predicted from associated increases in water temperature. Marked increases in pH were apparent for all months except December and January. The net effect was an insignificant increase in the winter mean and a significant increase in the summer mean. Suspended solids showed no significant changes, with the exception of a rise in the month of February associated with the influence of a single storm event on the computed average. Finally, nitrogen showed little response to drought. As with dissolved oxygen, the pattern undoubtedly reflects anthropogenic influences on the pattern and rate of nitrate loading in the Thames catchment associated with agricultural and sewage treatment practices.

Sample data used to estimate each of the trend models are plotted as dotted lines in Figs. 2 and 3, with the estimated models plotted as solid lines. Outliers eliminated from the estimation data sets for Fig. 3 are plotted as solid squares (■). Chow tests indicated that all models, with the exception of the drought temperature model, were sensitive to changes in the data set used for estimation. Results of the tests are given in Table 3, together with the results of the residual normality testing, normalized version of the Shapiro–Wilk W statistic and runs test statistic. Reported values provide no evidence for questioning the statistical adequacy of the estimated models. Standardized residuals plots confirmed that residuals were homoscedastic.

Comparisons of the predrought and drought models in terms of significant model coefficients and proportion of explained variation are presented in Table 4. Variables entering into models as significant regressors are indicated with a square (■ for predrought, □ for drought). The results are indicative of structural change in the underlying function defining the relationship amongst modelled variables and therefore suggestive of a significant effect of drought on measured water quality parameters.

DISCUSSION

The extensive data set constructed for the Thames estuary has allowed a detailed comparison of water quality within the estuary during drought

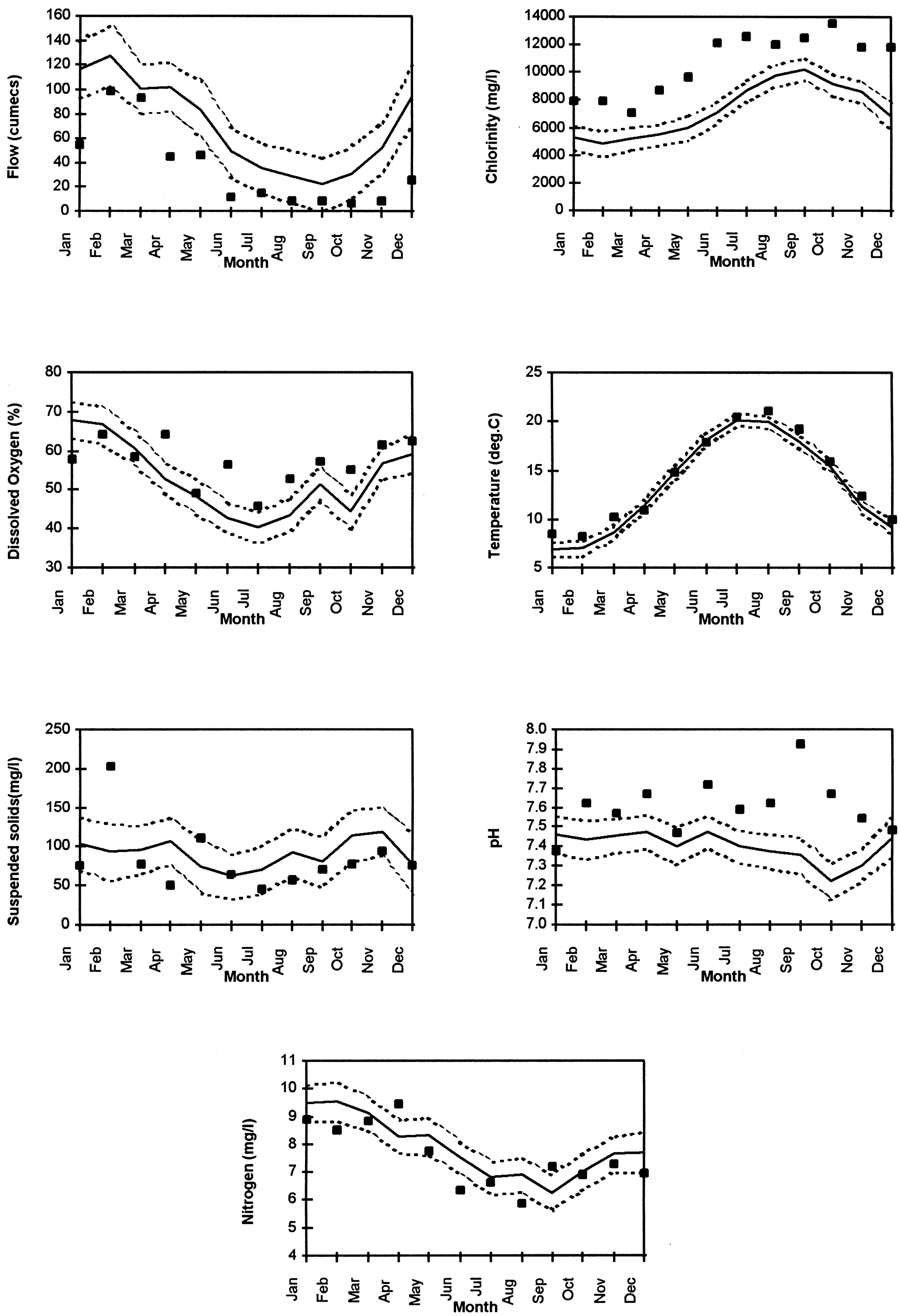

Fig. 1. Mean monthly values for water quality parameters during predrought and drought periods. Solid line = 1977–1988 means, dotted line = 95% confidence intervals, ■ = drought monthly means.

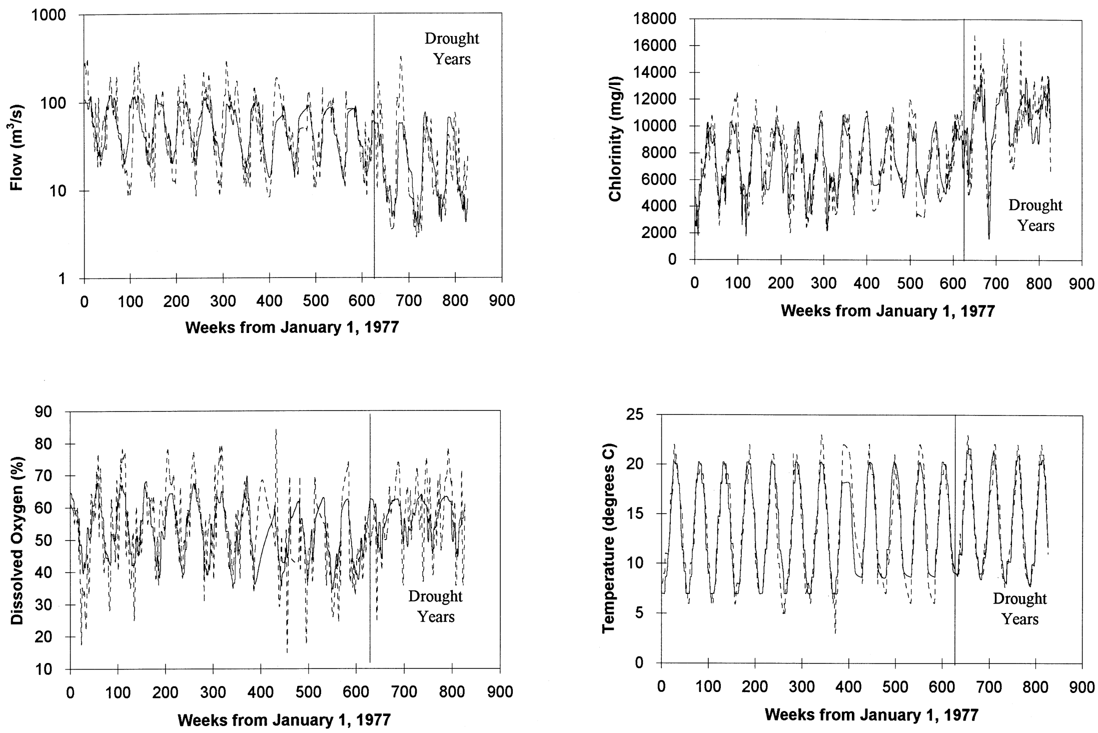

Fig. 2. Plots of model results for freshwater flow, chlorinity, dissolved oxygen and temperature, 1977–1988 (predrought period) and 1989–1992 (drought period). Dotted line = actual values, solid line = model.

and non-drought periods, the drought years selected (1989–1992) being the driest on record (Littlewood and Marsh, 1996). For all water quality parameters investigated (except nitrogen), drought conditions resulted in a significant change in mean values during summer, with several parameters also varying during the winter period. Additionally, none of the seven significant models constructed for the available water quality parameters under normal flow conditions fitted the data during drought years, the fluctuations during 1989–92 being described by alternative models (often with a different set of significant variables and temporal trend).

The drought conditions are reflected, as expected, in the Mann–Whitney test results and the model for freshwater flow entering the estuary. For the whole of the drought period, mean flows were significantly lower than the predrought years, regardless of season. This is highlighted by the model, with most maxima and all minima post-1988 being markedly lower than in predrought years. There was also a shift in the significant seasonal variables contributing to the flow models, with the important months during drought years being at the end of the year (September–December). The inclusion of these months reflects the continuation of summer low rainfall conditions through the autumn period and a shift in the time period when increased amounts of precipitation fell in the catchment (Attrill *et al.*, 1996). In addition to being exceptionally dry, the drought period was characterized by comparatively high temperatures. The influence of flow and temperature, respectively, on the measured concentrations and solubility of other water quality determinants during the drought is reflected by the fact that one, or both, of the parameters feature in the fitted models describing fluctuations in water quality during the drought period.

The implications of changes in freshwater flows for estuaries are not fully understood. Changes in flow are known to affect the composition of phytoplankton species and the factors responsible for phytoplankton growth (e.g. Marshall and Alden, 1997). Flow changes are less well understood for zooplankton or the larval fish communities that routinely exploit estuaries as nursery habitats. Although it is known that fish larvae are retained in some estuaries by counterflowing tidal and riverine currents (Graham, 1972), and that changes in freshwater flow undoubtedly affect larval retention and year-class strength, much work remains to be done before a comprehensive understanding of fluctuations in freshwater flows on pelagic estuarine fauna is achieved (Drinkwater, 1986).

Temperature increases during the drought period were more or less evenly distributed throughout the year, although increases during the winter months (+1.63°C) did exceed those occurring during the summer months (+0.90°C). Average temperature increases resulted from generally higher minimums

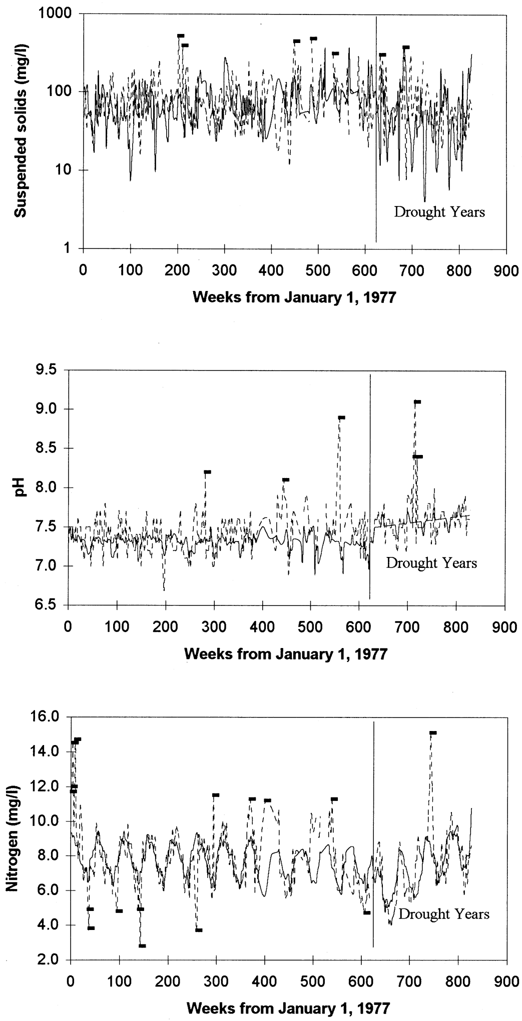

Fig. 3. Plots of model results for suspended solids, pH and nitrogen, 1977–1988 (predrought period) and 1989–1992 (drought period). Dotted line = actual values, solid line = model, ■ = outlier.

(7°C) during the drought as compared to the predrought (3°C) period. Temperature has been shown to have specific effects on the fish community. Average egg incubation periods for marine and estuarine fishes are lowered as temperatures increase (Rombough, 1997). Larval fish metabolic and growth rates increase with temperature (Houde, 1989), but when viewed across a full range of species, temperature seems to have little effect on larval recruitment (Rombough, 1997). The apparent contradiction has led some to speculate that temperature increases are associated with higher instantaneous mortality rates that are offset by better growth (Pepin, 1991). The net effect of the drought on estuarine dependent species of fish may, therefore, be negligible. Temperature changes can, however, trigger other important changes in the structure of estuarine fish communities. Holmes and Henderson (1990) report dramatic

Table 3. Results of diagnostic tests for fitted models. Values of the normalized Shapiro–Wilk or runs test Z statistic > 1.96 or < −1.96 indicate nonnormal or nonrandom residuals respectively. The Chow test is used to establish the stability of estimated model coefficients when the estimation data set is expanded. Tests here used the estimated predrought and drought models respectively and expanded their data estimation sets to include the data from the other period. Values italic indicate the new data had no significant effect on estimated model coefficients at the $a = 0.05$ level of significance

Model	Shapiro–Wilk normalised W statistic	Predrought Runs test Z statistic	Chow test F statistic	Shapiro–Wilk normalised W statistic	Drought Runs test Z statistic	Chow test F statistic
Flow	0.628	−1.716	4.65	0.796	−0.725	24.71
Chlorinity	−0.972	−1.409	13.45	0.623	−1.268	20.47
Temperature	−0.234	−0.708	13.58	−0.308	0.906	*1.87*
Dissolved oxygen	−1.743	−1.560	8.60	1.895	−1.811	183.75
PH	0.926	−1.488	3.50	−1.187	−0.482	9.08
Suspended solids	0.372	1.063	5.65	−1.103	−1.780	5.56
Nitrogen	1.301	−1.624	14.24	1.447	−0.739	14.68

Table 4. Seasonal and physico-chemical variables featuring in predrought and drought fitted models for water quality parameters, together with r^2 for each model. Seasonal variables: J–D = January–December, physico-chemical variables: CH = chlorinity, FL = flow, TE = temperature, SS = suspended solids, NI = nitrogen, WK = weeks

Variable	Seasonal Variables												Physico-chemical variables						r^2
	J	F	M	A	M	J	J	A	S	O	N	D	CH	FL	TE	SS	NI	WK	
Chlorinity																			
Predrought			■	■	■	■	■				■			■	■				0.63
Drought										□		□		□	□				0.52
Flow																			
Predrought				■				■	■	■	■				■			■	0.48
Drought									□	□	□	□			□				0.57
Temperature																			
Predrought			■	■	■	■	■	■	■	■	■	■							0.90
Drought	□	□			□	□	□	□	□	□	□							□	0.92
Nitrogen																			
Predrought									■					■	■			■	0.41
Drought														□	□			□	0.53
Dissolved oxygen																			
Predrought									■						■			■	0.45
Drought					□		□								□				0.25
Suspended solids																			
Predrought							■	■	■	■				■				■	0.96
Drought							□	□	□	□	□			□					0.92
pH																			
Predrought										■	■		■			■	■	■	0.81
Drought	□					□												□	0.94

increases in the abundance of numerous species in the estuary of the Severn during the abnormally warm winter of 1989 with warmer water species like red mullet (*Mullus surmuletus*) and sea bass (*Dicentrarchus labrax*) rising in relative abundance. Higher temperatures may also influence the movement of migratory fish, such as salmon (*Salmo salar*), particularly when coupled with low freshwater flows (Tinsley, 1998).

As would be expected, chlorinity levels within the estuary were strongly linked with flow, the position of salinity bands within the estuary being determined over the long term by the flow regime (Attrill *et al*., 1996; Kurup *et al*., 1998). As a result, maximum and minimum chlorinity levels during the drought were 1.34 and 2.05 times larger than the extremes achieved during the predrought period. A notable feature of the chlorinity pattern during drought years was a breakdown in the strong annual periodicity due to reduced winter rainfall. This is reflected in a reduction in the range of seasonal variables featuring in the chlorinity model. Under the predrought flow regime the spring/early summer months (March–July) were significant variables as during this time a predictable reduction in flows, and thus increase in chlorinity, was occurring. However, due to low rainfall in winter (particularly during 1991–92) this seasonal increase in chlorinity was not so consistent in and so these months did not feature in the drought model.

A reduction in oxygen levels during summer is a standard feature of estuarine water quality (Breitburg *et al*., 1997; Kinniburgh, 1998) due to a combination of low flows and, more particularly, increased temperature enhancing bacterial activity. It could therefore be anticipated that a further increase in temperature and additional flow reductions due to drought would have serious consequences for oxygen levels, but this was not the case in the Thames Estuary. Dissolved oxygen levels were significantly different in summer between drought and predrought years, but concentrations were higher during the drought period. The fitted model for oxygen indicates a decrease in the amplitude of fluctuations in concentrations, due mainly to a reduction in the number of low oxygen records post-1988. Oxygen sags were a feature of the estuary during summer for an extensive period during its rehabilitation (Wood, 1980) and have continued to pose a severe problem to water quality managers due to the run off from overflow drains during storm events (Ellis, 1989). These discharge untreated sewage and road run-off directly into the Thames estuary, adding to the BOD burden and promoting bacterial activity (Lloyd and Cockburn, 1983; Kinniburgh, 1998). In severe cases, these sags have had to be treated using a mobile oxygen injection system (Griffiths and Lloyd, 1985). It is therefore possible that during drought conditions, a reduction in the number of storm events resulted in less untreated sewage material entering the estuary and therefore fewer, or less severe, oxygen sags occurring during the summer. The relatively high oxygen levels may also be a reflection of the improved effluent quality of major sewage treatment works bordering the estuary during the drought period (Kinniburgh, 1998) and the increasing proportional importance of anthropogenic inputs to determining overall water quality during low flow periods.

Summer levels of suspended solids were significantly lower during drought years. This is not unexpected as suspended solids loads tend to increase with flow (Alden, 1997), the lower flows in drought periods (particularly summer) reducing suspended solid input from riverine sources and storm outfalls and allowing the settlement of suspended material in the estuary. This importance of flow as a controlling factor for suspended solids was reinforced by the models constructed for this parameter, with flow being the only significant physico-chemical variable featuring in both predrought and drought models. A reduction in suspended solids load during drought conditions may have consequences for estuarine organisms, particularly algae which, in upper estuaries, may be more productive due to the increase in light penetration resulting from lower turbidity (MacIntyre and Cullen, 1996; Irigoien and Castel, 1997). In the upper Thames estuary, increases in phytoplankton production have resulted in super-saturation of dissolved oxygen (Kinniburgh, 1998). However, it is unlikely that increased photosynthesis is an explanation for drought-related increases in oxygen levels in the mid-estuary as phytoplankton biomass decreases rapidly with increasing salinity and turbidity, with very little production occurring in the area off West Thurrock (Kinniburgh, 1998). Higher levels of turbidity can also decrease predation risks for juvenile marine fishes using the estuary as a nursery area (Blaber and Blaber, 1980), so it is possible that drought-induced reductions in suspended solids may increase natural mortality and reduce year-class strength.

Of all the measured water quality parameters, pH demonstrated the greatest changes between predrought and drought periods. Values of pH were significantly higher under drought conditions for both summer and winter, and the comparative models for the two periods showed the greatest disparity of the seven constructed. During predrought conditions, chlorinity, suspended solids and nitrogen concentrations all featured as significant variables. None of the variables were significant post-1988. In addition the model featured different seasonal variables. The input and composition of freshwater has been noted as being influential in determining the pH of estuarine waters (Lebel *et al.*, 1983), freshwater tending to have a lower pH than seawater. Whilst changes in pH of estuarine systems have been recorded as having an impact on certain organisms (Sunitha and Jayaprakas, 1997), any change in pH is likely to be more influential by acting indirectly. Decreases in pH can potentially release both heavy metals (Bueno *et al.*, 1998) and nutrients (De Laune *et al.*, 1981; McComb *et al.*, 1998) from estuarine sediments with consequences to flora and fauna (e.g. Seitzinger, 1981). Although metal levels in Thames estuary water have decreased dramatically over the last decade (Power *et al.*, 1999b), a large reservoir of metals remains in the sediment (Attrill and Thomas, 1995) and the overall potential impact of this source of metal exposure in the face of increases in pH is difficult to predict. Increases in pH are known to increase cadmium and zinc toxicity (Sorensen, 1991), but similar effects are not obtained for copper where increases in pH are generally thought to render fish more resistant to copper exposure (Sorensen, 1991).

Total nitrogen concentrations did not differ significantly between predrought and drought years, although the fitted model for the drought period did lose its seasonal component, perhaps a reflection in the changing temporal flow regime. Several nitrogen "spikes" were apparent, probably as a result of storm discharges and land run-off. Flow and temperature were consistently important factors influencing nitrogen concentrations, but lower flows did not alter the overall nitrogen levels in the estuary at this point. Nitrogen levels tend to decrease markedly from river to sea (Kinniburgh, 1998), so a reduction in riverine input from agricultural run-off may have a greater effect in the upper reaches of the estuary. Additionally, the major sewage treatment works for the London region discharge into the mid-estuary (Andrews, 1980) and the constancy of their input during drought periods may be influential in maintaining nitrogen levels during lower freshwater flows. From the results of this study, however, and in contrast to other water quality parameters, drought conditions do not appear to dramatically influence mid-estuary nitrogen concentrations.

CONCLUSIONS

This study has demonstrated that drought conditions can have significant effects on the water quality of estuaries. Models have indicated that under both drought and predrought conditions, most water quality parameters are influenced by a combination of freshwater flow and temperature. Reductions in freshwater flow entering the estuary, due to natural lowering of water levels in rivers and increased abstraction, therefore have a major influence on the quality of estuarine water many kilometers beyond the point of entry. Due to the close relationship between flow and salinity, it could be argued that water quality in estuaries is more vulnerable to reductions in flow than similar characterizations of rivers. Reductions in flow lead to measurable and statistically significant changes in estuarine water quality. Amongst the most affected variables is the key controlling parameter of salinity. It is suggested, therefore, that water management policies designed to assess and alleviate the effects of drought on river systems include consideration of the wider consequences of their actions because of the implications they may hold for estuarine water quality.

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