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## The evolution of water quality criteria in the United States, 1922–2003

**Alfred Dufour and Stephen Schaub**

*National Exposure Research Laboratory, US Environmental Protection Agency, Cincinnati, OH, and Office of Water, US Environmental Protection Agency, Washington, DC*

The microbiological quality of recreational waters was first discussed in the United States as early as 1922 by the American Public Health Association's Committee on Bathing Beaches (APHA, 1922). The Committee surveyed 2000 physicians and state health officials inquiring about the prevalence of infections associated with bathing places. The Committee Report in 1924 (APHA, 1924) reviewed the survey results and concluded that there was not enough evidence to develop bathing water standards for natural waters. In June 1933 the Joint Committee on Bathing Places was formed and in their first report noted that because of the great lack of epidemiological information no bacterial standards were adopted (APHA, 1933). They also stated that they did not want to propose arbitrary standards or measures that might promote public hysteria about the dangers of outdoor bathing places. By 1936 they were still not convinced that bathing places were a major health problem and re-stated their position on developing bacterial standards for bathing places.

The reluctance to propose bacterial standards for outdoor bathing places was again evident in 1936, 1940, and 1955 (APHA, 1936, 1940, 1957). The Committee did attempt to find evidence of the risk of illness from bathing waters prior to their reports in 1940 and 1955, but they found no compelling evidence. They stated that very little reliable data were available to implicate bathing places in the spread of disease (APHA, 1957).

Various states, meanwhile, began to put in place water quality criteria for their bathing places. These criteria ranged from 50 coliforms per 100 ml to 2400 coliforms per 100 ml.<sup>1</sup> The great diversity in state standards for bathing beach waters likely resulted from a lack of guidance from federal authorities or national organizations with interests in this area. In 1963, the Sanitary Engineering Division of the American Society of Civil Engineers, in a progress report (Senn, 1963), included an appendix on the then current state standards for the bacteriological quality of surface waters used for bathing, compiled by William F. Garber. This list is very informative on the various standards and approaches taken to monitor the quality of bathing beach waters. It includes a survey of all 50 states, the standard they used for controlling surface water quality, how the standard was calculated, and whether they used a sanitary survey in conjunction with their standard. Only 12 states had no standard for bathing beach waters in 1963. The other 38 states had standards which differed significantly from each other.

Among those states that did have microbiological standards for water quality, all used coliforms as the measure of that quality and all used the most probable number (MPN) procedure to estimate the number of coliform bacteria in a water sample. The number of coliforms used as a limiting value for water quality ranged from 50 per 100 ml to 2400 per 100 ml, with about half of the states using 1000 coliforms per 100 ml as the limiting value. Over half of the states calculated the upper limit using the arithmetic average of the MPN estimates. Approximately 23 % of the states used either the geometric mean or the median to calculate the threshold value. About 13 % of the states used a single threshold value, such as a percentile value.

Some interesting facts emerge from this listing of state bacteriological water quality standards that were in place in 1963, the most significant of which was the fact that they were all put in place without the benefit of health data showing a relationship between health effects in swimmers and water quality. At that time little information was available from epidemiology studies or from waterborne outbreaks of disease or case studies. Another interesting fact was the broad use of the arithmetic mean as a water quality standard prior to 1963, in contrast to the almost universal use of the geometric mean today. The use of the arithmetic mean at that time may have been due to suggestions by Thomas (1955) and Pomeroy

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<sup>1</sup> The term 'coliform' was used in the referenced papers to describe bacteria that were believed to be always associated with fecal contamination. The term was changed to 'total coliform' around 1965 (Orland, 1965) when 'fecal coliform' was introduced for a subset of the coliforms. The fecal coliforms were thought to be more specific to fecal pollution. *Escherichia coli*, one of a few of the microorganisms that comprise the fecal coliform group, is used today because it is even more specific for identifying the presence of feces in water. Hereafter, in this chapter, the term 'coliforms' shall imply what we currently call 'total coliforms'.

(1955). Thomas indicated that the arithmetic mean of a time series of coliform density determinations is superior to other commonly used averages, such as the geometric mean and the median, as a measure of health hazards associated with enteric pathogens such as *Salmonella* and *Shigella*. Thomas' suggestion was based on the premise that the highest risk of infection was related to the highest values of coliform measured and that the median and geometric mean had a tendency to minimize the high values, resulting in an underestimate of the true risk. Discussions about the use of the geometric mean versus the arithmetic mean continue to the present day with regard to the setting a standards for recreational waters (Parkhurst, 1998). The current use of the geometric mean to set standards for water quality is based mainly on the fact that most environmental water quality measurements, after transformation to logarithms, have a normal distribution and that this normal distribution lends itself to the use parametric statistics for evaluating data.

Most of the states established their water quality standards on an individual basis, while other states used approaches developed by other states. There was no distinct pattern to the development of standards. Three examples of the diversity of setting water quality standards are given by the states of Connecticut, the Ohio River Valley Sanitation Commission, located in Cincinnati, Ohio, and California.

Connecticut surveyed their entire Long Island Sound coastline, taking samples every 1000 ft. Four classes of water quality were developed from the data. Class A included 0–50 coliforms per 100 ml. Class B included coliform densities from 51 to 500 per 100 ml, and Class C ranged from greater than 500 to 1000 coliforms per 100 ml. Class D included coliform densities greater than 1000 per 100 ml. Analysis of the miles of coastline that contain coliform densities less than 1000 per 100 ml indicated that this level could be obtained about 90% of the time (Scott, 1932). This classification of waters also correlated very well with the sanitary survey conducted during the shoreline water survey. Connecticut used the information to set a water quality standard of less than 1000 coliform per 100 ml. Water samples containing less than 1000 coliforms per 100 ml were graded as acceptable. This approach might be looked on as the attainment approach.

Streeter (1951), an engineer associated with the Ohio River Sanitation Commission, used an analytical approach to develop water quality criteria. He used data reported by Kehr and Butterfield (1943) who showed the relationship between typhoid morbidity and the ratio of coliforms to *E. typhosa* (*S. typhosa*). For morbidity rates in the Ohio River Valley the estimated ratio was about one *S. typhosa* to 170 000 coliforms. In addition to this ratio, Streeter used the number of bathers per day (1000 in his calculations), the estimated volume of water swallowed by a swimmer (10 ml) and the average coliform content of the bathing water per milliliter. His calculations indicated that over a 90-day swimming season, if a bather swam every day, his risk of exposure to one *S. typhosa* would-be approximately 1 in 19. Kerr and Butterfield had estimated that about one in every 50 persons who

ingested a single *S. typhosa* bacterium would actually contract typhoid fever, and this was used to estimate that over a 90-day season the risk would be about 1 in 950 exposures for an individual. Streeter considered this to be a very remote hazard. He concluded that water meeting a bathing water standard of 1000 coliforms per 100 ml would not present a great hazard to a single bather or group of bathers.

California's Bureau of Sanitary Engineering (1943) used a somewhat different approach to setting standards for marine bathing waters. A limit of 10 *E. coli* per milliliter was considered to assure safety for recreational use of surf waters. This standard was based on a number of considerations. First, the limit was about 500 times greater than the treated drinking water standard, which was well accepted at that time. Second, there were no epidemiological studies to indicate that waters within this standard were associated with health effects. Third, natural bodies of water seldom exceeded this level. Fourth, the limit seems to be based on common sense and therefore acceptable to the public. It is perhaps coincidental that these three states, even though they used different approaches in the development of bathing water standards, all arrived at the same unacceptable level of bacteriological water quality, namely, 1000 coliforms per 100 ml of sample for both fresh and marine waters.

Much of the reluctance on the part of state and local authorities to set bathing water quality standards was due to the lack of evidence concerning the level of microbial pollution that was detrimental to the health of bathers. In order to remedy this lack of information, the US Public Health Service (PHS), in 1948, initiated a series of studies to obtain data on the relationship between bathing water quality and illness among swimmers (Stevenson, 1953). The first of three studies was conducted at two Lake Michigan beaches in Chicago. The investigators were unable to show an excess illness in swimmers at either of the beaches when the median coliform density was equal to or less than 190 per 100 ml. Further analysis was performed on selected data from both beaches. The three highest and three lowest coliform values were selected from each beach to determine if there was a difference in the gastrointestinal illness incidence rates. A statistically significant difference in the illness incidence rate was observed only at one beach where the mean coliform density from the three high days was 2300 per 100 ml. and the coliform density from the three low days was 43 per 100 ml. The difference in the illness rate from the second beach, where the mean coliform density from the three high days was 730 per 100 ml, was not great enough to rule out the possibility that it may have resulted from chance. The second study was conducted at a beach in Dayton, Kentucky, on the Ohio River near Cincinnati, Ohio. In this study an excess of gastrointestinal illness in swimmers was observed when the median coliform density was about 2300 per 100 ml. The third study conducted by the PHS was at two marine beaches on Long Island Sound, one at New Rochelle, New York, and the other at Mamaroneck, New York. The median density of coliforms in the New Rochelle Beach waters during the study was 610 per 100 ml, while at the Mamaroneck beach the median coliform density was 253 per 100 ml. No excess gastrointestinal illness in swimmers was observed at either beach.

The results from these studies suggested that swimming-associated gastroenteritis was not evident until the quality of the bathing water exceeded a median coliform density of at least 2300 per 100 ml. The results indicate that most of the states participating in the Garber survey in 1963 had set very conservative water quality standards.

In 1968 the Federal Water Pollution Control Administration commissioned the National Technical Advisory Committee (NTAC, 1968) to examine water quality criteria for a number of water use issues, including the relationship between swimming-associated illness and water quality. The Committee discussed the shortcomings of coliforms as an indicator of the presence of fecal contamination and declared that fecal coliforms were a much more specific indicator of fecal contamination. A study conducted on the Ohio River showed that fecal coliforms comprised about 18 % of the coliforms. This ratio was applied to the coliform density observed in the PHS studies where swimming-associated health effects were detected when the coliform density was about 2300 coliforms per 100 ml. The resulting equivalent fecal coliform density was approximately 400 fecal coliforms per 100 ml. The Committee suggested that a safety factor should be included in the new guideline, such that the water quality would be better than that at which a gastrointestinal health effect occurred, i.e., 400 fecal coliforms per 100 ml. The recommendation of the Committee was:

Fecal coliforms should be used as the indicator organism for evaluating the microbiological suitability of recreational waters. As determined by multiple-tube fermentation or membrane filter procedures and based on a minimum of not less than five samples taken over not more than a 30-day period, the fecal coliform content of primary contact recreation waters shall not exceed a log mean of 200/100 ml, nor shall more than 10 percent of total samples during any 30-day period exceed 400/100 ml.

The NTAC recommendation was, in part, based on two unsupported assumptions. The first was that the fecal coliform portion of the coliform population was constant in all waterbodies of the United States, including marine waters. The data used by the NTAC had been obtained in the early 1960s from water samples collected from the Ohio River, which at that time received treated and untreated fecal wastes from many sources, not all of which were necessarily representative of sources across the United States. The second assumption, and possibly the more significant of the two, was the belief that by halving the indicator density at which a detectable health effect occurred, namely from 400 to 200 fecal coliforms, a zero risk would result. This assumption is considered unreasonable because the relationship between swimming-associated illness and water quality cannot be calculated from a single point (2300 coliforms per 100 ml). Both of these assumptions would have significant effects on later efforts to develop criteria and guidelines for the quality of recreational waters.

The potential for gastrointestinal illness to be associated with exposure to natural bathing waters was considered again in 1972 by the US Environmental

Protection Agency (EPA). Two series of studies were conducted, one at marine beaches (Cabelli, 1983) and the other at freshwater beaches (Dufour, 1984). The marine beaches were located in New York City at Coney Island and Rockaways beaches, in Boston at Revere and Nahant beaches, and at Lake Pontchartrain in the New Orleans area. The studies were conducted over a six-year period. Analysis of the data from the three locations shows that there was a linear relationship between the rate of swimming-associated gastroenteritis and the quality of the water measured with enterococci using the membrane filter technique but not so for *E. coli*. The swimming-associated illness rates ranged from zero per 1000 swimmers to 28 per 1000 swimmers with a mean rate of 15 illnesses per 1000. The water quality levels ranged from 3 enterococci per 100 ml to about 500 enterococci per 100 ml.

The freshwater studies were conducted at locations in Erie, Pennsylvania, on Lake Erie and at Keystone Lake about 60 miles east of Tulsa, Oklahoma. The studies took place in 1979, 1980, and 1982. The average rate of swimming-associated gastroenteritis was about 6 per 1000 swimmers with a high rate of 14 per 1000 swimmers and a low rate of zero per 1000. The mean water quality level, as measured with *E. coli*, was 72 per 100 ml and the mean level for enterococci was about 20 per 100 ml. The enterococci densities ranged from 6 to 80 per 100 ml. *E. coli* densities ranged from 18 to 250 per 100 ml. There were no significant illness correlations with other fecal indicators used in these studies.

In 1986, new criteria for recreational waters were recommended by the EPA, based on data collected during the above marine and freshwater studies (Federal Register, 1986). The recommendations were, in marine waters:

A geometric mean of 5 samples taken at equal time intervals over a 30-day period shall not exceed 35 enterococci per 100 ml.

and in freshwaters:

A geometric mean of 5 samples taken at equal intervals over a 30-day period shall not exceed 126 *E. coli* per 100 ml, and

A geometric mean of 5 samples taken at equal time intervals over a 30-day period shall not exceed 33 enterococci per 100 ml.

The upper threshold limit of enterococci in marine waters represents an acceptable risk of 19 swimming-associated illnesses per 1000 swimmers, while the upper limit for *E. coli* or enterococci in freshwaters represents an acceptable risk level of 8 gastrointestinal illnesses per 1000 swimmers. The basis for these acceptable risk levels is found in the EPA's desire to develop risk levels no more or less stringent than what was accepted under the fecal coliform limit of 200 fecal coliform per 100 ml. Although it was commonly believed that the 200 fecal coliform limit represented a zero risk, the EPA studies cited above clearly showed that this was not true.

The risk level for marine waters was based on three known values, all obtained from data collected during the epidemiological studies at marine bathing beaches. They included the mean of the enterococci values per 100 ml from all of the beaches studied, a similar mean value for fecal coliforms and the recommended fecal coliform criteria of 200 per 100 ml. These values were used in the equation  $A = (C \times B)/D$ , where  $A$  is the unknown value of the criterion for enterococci,  $B$  is 29, the overall mean of enterococci,  $C$  is 200, the criterion for fecal coliform, and  $D$  is 166, the mean of fecal coliform from the marine beach epidemiological studies. Solving the equation gives 33 enterococci per 100 ml, the currently recommended criterion for marine waters; similar calculations were made for the freshwater criterion using a fecal coliform mean of 115 per 100 ml and an enterococci mean of 20 per 100 ml or an *E. coli* mean of 72 per 100 ml. The gastrointestinal illness rates of 8 and 19 per 1000 swimmers can be calculated using the regression equations found in EPA (1986). This ambient water quality criteria document also indicated single-sample maximum (SSM) values that might be used by states for other less frequent exposures to water than full body contact at designated beaches. These SSM values are based upon the upper 75th percentile values for designated beach waters. The numbers have been established for both enterococci (62 per 100 ml) and *E. coli* (235 per 100 ml) in fresh water and enterococci (104 per 100 ml) in marine waters. The SSM values were set at the 82th percentile level for moderate full body contact recreation, at the 90th percentile value for lightly used full body contact and at the 95th percentile level for infrequently used full body contact. The SSM is viewed by EPA as a useful measure for making beach notification and closure decisions, especially in situations where sampling is infrequent, e.g., once a week. At some point in time many beach managers began monitoring their waters on a daily basis and the SSM allowed them to make decisions based on the daily sample.

The methods chosen for measuring fecal indicators are as important as the selection of indicators used for measuring water quality, since the methods govern how accurate, precise, and specific the water quality measurements are. The goal in 1915 of water microbiologists was to be able to quantify the presence of fecal contamination using microbes that were constantly associated with feces. The technology for growing fecal associated organisms was very crude and, in the case of coliforms, the most effective means of detecting these organisms was in a tube of liquid culture medium containing lactose. The lactose was readily fermented by coliforms and in the process produced large amounts of gas. The gas produced by coliforms in the liquid medium was captured in a second small tube inverted in the culture tube. The captured gas was easy to visualize and indicated that the tube contained growing coliforms. McCrady (1915) laid the foundation for the application of probability theory to the bacteriological examination of water by the fermentation tube method. His method, which was later improved by others, was the key to estimating the density of coliforms in water. This method for estimating the number of coliforms is called the most probable number (MPN) method and has been in use for almost 100 years. The method does have some drawbacks. It

is very imprecise, it somewhat overestimates the true density of conforms in the water, and it cannot easily accommodate large volumes of sample.

In the early 1950s another method for measuring coliforms was introduced to water microbiologists by Goetz and Tsuneishi (1951). It made use of cellulose membranes, through which water samples could be passed. The membranes had a porosity which retained particles the size of bacteria. After filtration the membrane filters were then placed on growth medium, where, after a suitable incubation period, colonies of bacteria grew on the membrane. The colonies were easily visualized and the numbers of colonies per test volume of filtered sample could be calculated. Clark *et al.* (1951) described the use of membrane filtration (MF) for measuring coliforms. Some of the advantages of MF method, relative to the MPN method, were that it was more precise, it could accommodate larger water samples, the colony count results were more reproducible, and the assay system required less time to perform. MF disadvantages are that turbid waters, for example those contaminated with clay or algae, can foul the membranes and preclude the use of the test, and competition from high densities of non-target bacterial colonies can influence test results.

The PHS epidemiological studies were conducted using the MPN procedure for measuring water quality (Stevenson, 1953). The EPA epidemiological studies in the 1970s and early 1980s used MF methods to measure water quality (US EPA, 1986). Prior to 1960 all states used the MPN method to measure their water quality. In the NTAC (1968) report both the MPN and MF were recommended for measuring recreational water quality. The 1986 water quality criteria for recreational waters recommended only the MF method (Federal Register, 1986). It is not always clear in the beach survey literature what method is used by states to determine the quality of their waters, although most states do use the MF test. However, several states continue to use the MPN procedure.

In 1988, the EPA surveyed the 50 states to determine what standards they had in place (US EPA, 1988). Forty-six of the states had adopted the fecal coliform as the indicator of choice. Three states were using enterococci for marine waters and one state had selected *E. coli* for measuring the quality of freshwaters. Forty-three of the states used a geometric mean of 200 per 100 ml as a control limit and three used a single-sample fecal coliform limit of 200 per 100 ml. Four states used a median as the controlling limit. Three states used a control limit of fecal coliform less than the 200 per 100 ml limit. Three states used total coliforms as a second indicator, usually for freshwaters. Significant differences in the 1988 survey, relative to the 1963 survey conducted by Garber, were the indicator of choice and the calculation of the upper control limit value. In 1963, all 50 states used coliforms to measure water quality, whereas in 1988, 46 used fecal coliforms, the indicator recommended by NTAC (1968) and US EPA (1976). In 1988, 43 states had adopted the geometric mean to calculate the water quality value. In 1963, over half of the states used the arithmetic mean. The almost total conversion to the use of the geometric mean was more likely due to its recommendation by the NTAC and the U.S. EPA, although it appears that no formal discussion of why it should be used took place.

In 1992, the EPA published a report summarizing the water quality standards for bacteria that were then in force for each of the 50 states (US EPA, 1992). The listing indicated that for freshwaters, used for primary contact recreation, 43 states used fecal coliforms and the control limits ranged from 22 to 200 per 100 ml. Several states used values in the 2000 per 100 ml range. Seven states used either enterococci or *E. coli*, usually with an upper limit of 33 per 100 ml or 126 per 100 ml, respectively. Eighteen states developed or used marine water quality standards for fecal coliforms with upper control limits of 14 to 200 per 100 ml. Four states adopted enterococci or *E. coli* standards with upper limits of 7 to 35 for the former and 126 for the latter.

In 2003, the EPA updated the list of individual state bacterial water quality standards that were in place (US EPA, 2003). There was not much change in the distribution of the types of indicators used to measure water quality. In freshwaters there was a decrease from 43 to 40 in the number of states using a fecal coliform standard and for marine waters there was a decrease of six, from 18 states to 12, states that used fecal coliforms, and an increase in the use of enterococci for *E. coli* from four states to 11 states. The control values for enterococci and *E. coli* tended to follow the EPA recommended values; however, one or two states chose to put standards in place that were more stringent than the values recommended by the EPA. A more recent listing of state standards is not available, but it is likely that more states, especially coastal states, will adopt *E. coli* or enterococci as a result of the passage of the Beaches Environmental Assessment and Coastal Health Act in 2000, which encouraged states to adopt new standards based on the 1986 recommendations suggested by the EPA. There are some interesting findings in the latest listings of microbial standards used by states. First, there is still a great reluctance by many states to change their fecal coliform standards to ones that are more contemporary, for example, enterococci or *E. coli*. Second, there is some variability in the selection of upper control limits, indicating a degree of uncertainty by some states relative to which acceptable risk level is appropriate for primary contact recreational waters.

The 1986 based criteria, although they appear to be quite different from early water quality criteria, have threads that go back to the 1000 coliform per 100 ml standards in use before 1968. The PHS studies observed a detectable health effect when the coliform density was about 2300 per 100 ml (Stevenson, 1953). The NTAC (1968) translated the coliform level to 400 fecal coliforms based on a coliform to fecal coliform ratio, and then halved that number to 200 fecal coliforms (as a safety factor), which, in theory, was the equivalent of 1000 coliforms. In 1986 the EPA translated the 200 fecal coliform recommendation to *E. coli* and enterococci in order to maintain a risk level generally accepted prior to that time. What we have then, is 126 *E. coli* and 33–35 enterococci being equivalent to 200 fecal coliforms, and by extension equivalent to 1000 coliforms, a water quality standard in place in many states before any health studies were conducted. The reluctance to change health risk levels, and the associated bacterial densities they are related to, has made the development of recommendations or standards for recreational waters difficult

to reconcile with scientific findings. The different approaches used over the years to develop standards for recommended criteria has in part been due to the way in which data are interpreted and that interpretation has relied frequently on the contemporary statistical tools available at the time. Issues that have been debated historically are still discussed today. Notable among those issues is whether to use the arithmetic mean or the geometric mean to characterize fecal contaminated waters using microbial measures. Current practice favors the use of the geometric mean for characterizing the quality of recreational waters. Although this subject has surfaced from time to time, no thorough discussion of the issue has taken place. Another issue that is quite obvious, if one reviews the development of criteria and the setting of standards in past years, is acceptable risk. How to select an acceptable risk level has been a much debated subject in the last half-century and one that has never come to closure. For some zero risk is the only acceptable risk, while for others risks other than zero are quite acceptable. This problem is due to the fact that there is no straightforward process leading to a judgment about where an acceptable risk level should be set. These issues and others will be discussed, in part, in the chapters that follow, in an attempt to clarify and define the statistics and procedures for evaluating microbiological and health data collected during the conduct of epidemiological studies and the design of plans for monitoring beach waters.

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