Economic and Health Risk Trade-Offs of Swim Closures at a Lake Michigan Beach

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This paper presents a framework for analyzing the economic, health, and recreation implications of swim closures related to high fecal indicator bacteria (FIB) levels. The framework utilizes benefit transfer policy analysis to provide a practical procedure for estimating the effectiveness of recreational water quality policies. Evaluation criteria include the rates of intended and unintended management outcomes, whether the chosen protocols generate closures with positive net economic benefits to swimmers, and the number of predicted illnesses the policy is able to prevent. We demonstrate the framework through a case study of a Lake Michigan freshwater beach using existing water quality and visitor data from 1998 to 2001. We find that a typical closure causes a net economic loss among would-be swimmers totaling \$1274−37 030/day, depending on the value assumptions used. Unnecessary closures, caused by high indicator variability and a 24-h time delay between when samples are taken and the management decision can be made, occurred on 14 (12%) out of 118 monitored summer days. Days with high FIB levels when the swim area is open are also common but do relatively little economic harm in comparison. Also, even if the closure policy could be implemented daily and perfectly without error, only about 42% of predicted illnesses would be avoided. These conclusions were sensitive to the relative values and risk preferences that swimmers have for recreation access and avoiding health effects, suggesting a need for further study of the impacts of recreational water quality policies on individuals.

Introduction

Fecal contamination of natural water bodies used for swimming is a significant public health concern worldwide. The need for practical and inexpensive risk screening methods has led to the use of fecal indicator bacteria (FIB) such as total coliform, enterococci, and *Escherichia coli* as a signal for unhealthy concentrations of human and livestock fecal waste in water. This approach avoids the need for costly direct testing for the many pathogenic bacteria, viruses, and protists associated with fecal waste and is based on numerous findings of association between FIB concentrations and elevated swimmer illness rates (*1−4*).

It has become common practice in the United States and elsewhere to use FIB testing as the basis for binary management choices about whether to impose temporary prohibitions on human contact with the water ("swim closures"). Nationwide, the number of natural swim areas being monitored is increasing, as is the frequency of swim closures (*5*). Although state laws and local implementation vary, the U.S. Environmental Protection Agency's (U.S. EPA) current recommended standard for freshwater is either a single sample of 235 *E. coli* colony-forming units (cfu)/100 mL or a geometric mean of 126 *E. coli* cfu/100 mL over five samples taken within the past 30 days (*6*). Typically, closures are implemented for the 24-h period following detection of an "exceedance", after which the jurisdiction may either end the closure or continue it until a subsequent water sample shows that FIB levels have returned to compliance.

At present, accepted and affordable FIB sample processing technologies require incubation for 18−24 h (*7*). Consequently, a swim closure cannot be instituted until the day *after* the water exceeded the standard. This makes the utility of swim closures highly dependent on the extent to which FIB levels are correlated from one day to the next. Recent studies at several freshwater beaches in Lake Michigan (*8−10*) and a marine beach in southern California (*11*) have shown that FIB levels can vary substantially over very small spatial (centimeters to meters) and temporal (minutes to hours) scales. These studies found little or no correlation between indicator levels from one day to the next. At Indiana Dunes State Park (IDSP) Beach, there is virtually no relationship between the *E. coli* level on days when samples were taken that exceeded the standard and the *E. coli* level on subsequent days when the results were reported and swim closures were instituted (Pearson $R^2 = 0.005$, $N = 45$) (*12*).

These findings call into question the potential effectiveness of current FIB-based swim closure practices. Since both monitoring programs and swim closures have real costs for coastal communities, it is important to explore the possible implications of this situation. This paper provides a framework for exploring the economic, health, and recreation implications of using a binary closure policy to regulate swim areas characterized by fluctuating and sporadically high FIB levels. We first describe and justify the use of transfer policy analysis to estimate the value of outdoor swim recreation, the value of different health states, and the epidemiology of swimming-related illnesses. We then present a four-stage analytical framework following a National Research Council protocol (*13*). Finally, we apply the framework in an assessment of a closure policy at a Lake Michigan freshwater beach using existing data from 1998 to 2001 and test the sensitivity of our methods and conclusions to a range of assumptions. The discussion summarizes and qualifies our findings and highlights key areas needing further research.

Transfer Policy Analysis

Transfer policy analysis refers to the set of techniques whereby economic valuation, risk, cost, or impact predictions from an existing study or studies are applied to estimate similar factors related to a distinct policy problem at a different place and time (*14*). The process may involve either meta-analysis,

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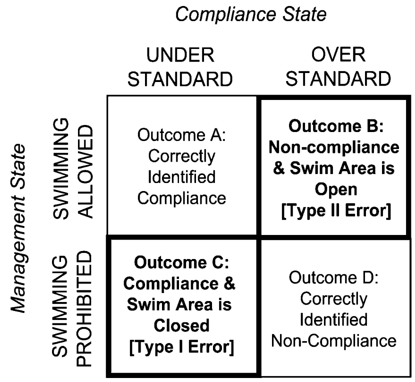

FIGURE 1. Typology of swim closure management outcomes in a binary choice setting. Closure decisions based on outdated information can result in either intended management outcomes (A and D) or unintended errors in the form of unnecessary (C) or missed (B) closure days.

if a large number of relevant original studies exist, or a combination of expert judgment and statistical extrapolation in situations where model or data inputs are scarce. Transfer is a cornerstone of practical policy analysis because of the widespread need for impact assessments throughout the policy development process despite time and budget constraints that prevent new site- and issue-specific studies (*15*).

The literature on transfer methods is well-developed and includes insights regarding its validity and limitations as well as the appropriate contexts in which it can be used. Three important conclusions emerge. First, the results from a transfer can only be as good as the original studies that are used. The valid use of transfer techniques demands attention to the same sources of error that affected the original studies (*16*). Second, the resources or services being valued and the user groups in the study and policy sites must be comparable (*17*). Third, different policy circumstances may permit different levels of accuracy in transfer estimates, depending on how the results will be used (*15*). Transfer analysis is particularly suitable for broad priority setting, understanding whether and where more precise information is needed, or developing a short list of feasible actions.

We assert that transfer is an appropriate and valuable analytical technique for assessing the economic and health trade-offs created by swim closures at specific beaches. Lack of information about the relative costs and benefits of swim closures has severely limited the formation of alternative monitoring and management strategies. Closures are also a highly local phenomenon; national-scale assessments are generally adequate for addressing local laws, practices, and ecological realities. Also, original site-specific valuation studies are likely to be financially and technically beyond the reach of local agencies in the short term, and a sufficient number of relevant studies of acceptable quality are available for transfer. We follow transfer methods used in past studies that relate to various components of our total problem (*18*, *19*).

Figure 1 illustrates the overall schema in which the transfer estimates will be applied. There are four management outcomes that result from the intersection of a binary swim closure policy with time-delayed indicator measurement. Two of these outcomes are consistent with the implied purpose of the policy: a swimming area is open when the water complies with the standard (outcome A) or closed if it exceeds the standard (outcome D). The other two outcomes are inconsistent with the policy and are consequently referred to as “closure errors”. These outcomes occur when a swim area is open while indicator levels actually exceed the standard (outcome B) or when swimming is prohibited despite the swim area being in compliance (outcome C). Outcomes B and C are analogous, respectively, to false negative (type II error) and false positive (type I error) test results in clinical diagnosis or scientific hypothesis testing. However, here we are referring only to the accuracy of the management outcome relative to the policy objectives and how errors are induced by time delay rather than the accuracy of the water quality test result relative to the true state of the world and how it is affected by the number of replicate samples taken. We will not explore the influence of the later source of error and instead assume for simplicity that the reported test result is the true population parameter. The null hypothesis is that the swim area is in compliance with the standard and should be open.

Four-Stage Transfer Procedure

Our approach follows a four-stage framework given by the National Research Council (*13*). The stages are hazard identification, risk assessment, valuation, and policy analysis. The first and second stages involve identifying and characterizing the risk for which FIB levels are meant to proxy. This is accomplished by identifying dose−response relationships from the literature that can be transferred using visitor and water quality data from the beach of interest. The third stage finds estimates for the dollar values that beach visitors place on swim recreation and avoidance of illness. Finally, the fourth stage presents a series of equations for applying and testing the importance of the transfer values in a comparison of management outcomes under a specific closure policy.

Stage 1: Hazard Identification. In this stage, the harmful agent(s) at issue and their critical features relative to the management problem are identified. The FIB of interest in the case study is *E. coli*, a bacterium naturally occurring in the digestive systems of healthy humans and many animals. As noted above, *E. coli* is the U.S. EPA recommended FIB for freshwater bodies and is consequently the type of FIB monitored at our case study site. Example sources of *E. coli* contamination in natural water bodies include pipe leaks, combined sewer overflows, boat dumping, animal droppings, or bather fecal accidents. The public health hazard is triggered when humans interact with and ingest contaminated water from a river, lake, or ocean.

It is important to note that this paper addresses *E. coli* only as an indicator for the presence of fecal material, not as a direct health hazard itself. This will limit our analysis to the types of moderate and short-term health effects associated with FIB in readily available epidemiologic studies. Chronic, serious disability, or fatality risks are not addressed. We feel these are necessary and reasonable assumptions for several reasons. First, not all *E. coli* strains are pathogenic, and typical *E. coli* testing procedures are not designed to discern the source(s) of the bacteria or whether toxic strains are present. Second, illness outbreaks attributed directly to recreational water borne *E. coli* in the United States are rare. For instance, of the 777 nationwide human *E. coli* O157:H7 infections confirmed by the Centers for Disease Control in the United States in 1998, only four of the illnesses and none of the deaths were tied to swimming (*20*). Finally, although detection of FIB may be associated with an increased chance of more serious fecal-related pathogens (e.g., hepatitis A, *Shigella*, *Cryptosporidium*, or Norwalk-like virus), we find no transferable studies that could be used to incorporate this effect.

Fundamentally, the pathogenic agent(s) truly affecting swimmer health at particular beaches are often unknown. The conclusions of a policy assessment are dependent on whether epidemiologic equations that accurately reflect the rates and typical severity of illnesses caused by the local water quality hazard are available. This in turn affects the selection of a health value parameter that accurately reflects the benefit of avoiding those precise effects. Absence of site-specific information about pathogen sources and their associations with fluctuating local densities of FIB and health impacts are principal sources of uncertainty. However, the approach of using FIB levels to estimate the rate of a limited type of swimmer illnesses is an accepted practice and the best available approach at this time.

Stage II: Risk Assessment. Ideally, an accurate characterization of the severity of the hazard requires current, locally collected information on three factors: information about the concentration of the indicator (occurrence), the number of persons using the swim area over a given period (exposure), and information about the dose−response relationship (a mathematical function relating a set of well-defined health effects to different levels of exposure to the indicator). Since no such local studies exist, we turn to the comprehensive literature review of Prüss (*3*), which confirms strong evidence of an association between the density of FIB in recreational waters and the rate of illnesses among swimmers as compared with nonswimmers at numerous freshwater and marine beaches.

For our case study, we use an equation from the same studies used to derive the current U.S. EPA guidelines in Dufour (*2*):

$$R = -11.74 + 9.397\log_{10}(E) \qquad (1)$$

where R is the swimming-associated gastrointestinal illness rate per 1000 swimmers and E is the recorded *E. coli* cfu/100 mL from water collected at the time of exposure. This relationship was derived from epidemiological studies at four North American freshwater beaches in which “swimming” was defined as exposure of all upper body orifices to the water for any length of time (*2*). Again, the structure of this equation implies that only the rate of swimmer illness increases as FIB density increases and not the severity of each illness case or the chance of experiencing more severe illnesses or death. At the current U.S. EPA guideline of 235 *E. coli* cfu/100 mL, approximately 10 swimmers out of 1000 will experience a gastrointestinal illness episode.

Several additional caveats should be noted. First, eq 1 was developed from studies in water bodies known to be contaminated with domestic human sewage waste (*2*). In contrast, source identification studies at our case study site have as yet failed to link local *E. coli* occurrence to human fecal point sources (*10*, *21*). There is evidence that the association between swimmer illness and high FIB density may be diminished or absent in bathing waters subject only to nonpoint sources (*22*). In addition, we simply lack the kind of high-resolution spatiotemporal data and epidemiologic observations to directly address all the possible health implications of spatiotemporal variability in FIB. We follow the common practice of using eq 1 to a make an estimate of the number of illnesses for an entire day based on a single-point FIB observation, despite the fact that FIB measurements are customarily taken at only one location and in the morning prior to warmer hours when most visitors probably swim. A recent Lake Michigan study showed that *E. coli* levels tended to decline significantly from morning to afternoon (*8*). These points suggest that eq 1 may overestimate the illness rate.

Stage III: Valuation. We assert that both swim recreation and the avoidance of potential health effects from pathogen exposure have positive economic value to all individuals who would voluntarily choose to swim. Determining the precise value of those services, however, is challenging because they are not directly traded or priced in any observable market. A significant body of literature spanning well over 20 years explores economic techniques for estimating the value people place on outdoor recreation activities such as swimming and on avoiding negative health changes. Dollars provide a useful common metric for expressing the relative value of nonmarket goods and services. Willingness to pay (or WTP), as measured in a carefully framed valuation survey, is a preferred nonmarket valuation metric because individuals tend to exaggerate when asked directly to place a value on an activity or a change in health without facing any tangible financial consequence (*14*). Quality WTP studies embed the valuation in a realistic context of relative change, a budget constraint, and other possible uses of funds. While different individuals and subgroups of the population may be affected differently from a similar activity or pathogen dose exposure, the WTP values reported here may be interpreted as mean responses of the public.

Walshet al. (*23*) conducted a meta-analysis of 120 outdoor recreation studies published between 1968 and 1988. Among the 11 identified studies that specifically addressed the value of swimming, the mean value per visitor day of swimming was \$22.97 in 1987 dollars. Adjusted to 2000 dollars using the Consumer Price Index (CPI), that value would be \$35.00 per visitor day. Rosenberger and Loomis (*24*) recently updated and expanded the analysis of Walsh et al. to include 170 recreation value studies, identifying a value for swimming of \$38.46 per visitor day in the northern United States in 2000 dollars. Finally, Sohngen et al. (*25*) reported a value of \$15.50 per visitor day for a beach visit in 1999 dollars (\$16.02 in 2000 dollars) on the basis of a survey of visitors to multiple Lake Erie sites. The consistency of reported values among different studies is reassuring as to their accuracy. We used the highest and lowest point estimates in order to test the influence of recreation values over a reasonable range.

Assignment of economic value to a change in health status ideally involves three elements: a clear definition of the health change (symptoms, duration, and effect on functional status); survey, interview, or actual cost data on how different individuals value the change; and translation of the results into a common metric, such as cost of illness, quality-adjusted life years, or dollars of WTP to avoid the change. We know of no study directly measuring the monetary value people place on avoiding illnesses due to swimming exposure. Thus, we surveyed the literature on the values people associate with avoiding food-borne illnesses similar to those caused by ingestion of the pathogens associated with high *E. coli* levels.

The closest analogous study that met our needs was Mauskopf and French (*26*), who report a per person WTP for government programs to avoid a mild case of gastrointestinal symptoms owing to salmonellosis (2−3 days) as \$280 and a moderate case (5−7 days) as \$1125, CPI-adjusted to 2000 dollars. This value reflects the total dollar amount that a typical adult individual would be willing to forego to avoid the cumulative impact of a multiday illness episode (case) encompassing many possible perceived effects, ranging from lost wages or added childcare expenses to pain and suffering. Again, we suggest using mild and moderate health effect values to test a range of amounts either lost or gained due to closure decisions is appropriate. Therefore, the policy comparison will be conducted under four different value assumptions sets, representing the four possible combinations of the higher and lower health and recreation value estimates.

Stage IV: Policy Analysis. The final stage consists of a series of equations that integrate and apply the risk and value transfer estimates. First, we calculate the per day net economic value to would-be swimmers of an open or closed swim area. Next, we show a flexible method for using historical data to calculate the daily probability of compliance states and management outcomes. Finally, we compare the net economic benefits of two policies: never closing the beach (hereafter no testing) and closing the beach when the previous day’s *E. coli* level exceeds the regulatory standard (hereafter testing). The testing policy represents an idealized version of the set of monitoring and closure practices commonly used at our case study site and numerous other beaches nationwide.

To explore the influence of inter-day variability in FIB levels on policy effectiveness, we conduct the policy comparison under two different scenarios. In the first, we assume no daily temporal correlation in FIB levels (non-informative scenario). In the second, we use historical records to establish the conditional probabilities of FIB levels changing day to day between the two states of being under (U) or over (O) the standard (informative scenario).

Per Day Net Benefits of a Management Outcome. The aggregate per day net benefit (B) to swimmers of any of the four management outcomes shown in Figure 1, given the management decision for that day, is estimated as

$$B = SFN\left[V_{\mathrm{recreation}} - \frac{RV_{\mathrm{health}}}{1000}\right] \quad (2)$$

where S is an indicator variable representing the management decision for that day and is equal to -1 or 1 if swimming is prohibited or permitted, respectively; F is the fraction of visitors that swim or would swim; N is the number of visitor days (i.e., visitors in a day); $V_{\mathrm{recreation}}$ is the value of swim recreation per visitor day; and V_{health} is the value of avoided health effects per visitor day. If swimming is prohibited, N may be estimated from an historical average. The technique of using total beach visitor statistics multiplied by a proportion that swim was chosen because readily available visitor statistics typically are not broken down by subactivity. Note that S acts as a scalar and that N and F equally affect both terms within the brackets, while the illness rate R (computed using eq 1 at the assumed or recorded FIB density for the day in question) and the recreation and health values are the primary drivers of the relative net benefit.

In formulating eq 2, we make no attempt to address economic effects that swim closures may have on nonswimming beach visitors or to other economic activities, such as concessions, local real estate markets, or regional tourism. While these effects are real for many communities, we limit our analysis to the swimming visitor population for simplicity to focus the assessment on the direct and personal impacts of closures and to preserve the generality of the framework. Also, we believe these assumptions make our assessment conservative. To our knowledge, there are also no estimates in the literature for a value that beach visitors who have no intention of swimming might place on the prohibition of swimming for other beach goers, for instance members of their own party. If we assumed a positive option or existence value for nonswimmers of a swim area being open, eq 2 would underestimate the losses associated with closed beaches.

Per Day Expected Net Benefits of a Policy. Results from eq 2 for typical days representing each management outcome can be combined with information about the likelihood of each outcome to estimate the per day expected net benefit of an overall testing and closure strategy. We adapt the approach of Olsen (*27*) for estimating the expected net benefits of no testing and testing under the two different FIB variability scenarios.

Define P_{u} as the proportion of days that the indicator level is below the regulatory standard based upon historical records over a specified period of time. Although P_{u} may take on different values at the same beach over time, we assume here that it is stationary over the chosen time period and can be estimated empirically. In locations where a sufficient historical testing record is unavailable, P_{u} may be approximated using data from an analogous location. In the absence of a testing and closure policy, only outcomes A and B are possible, and the expected net benefits per day of the no testing policy are simply weighted by the proportion of days spent in each state of either over or under the standard:

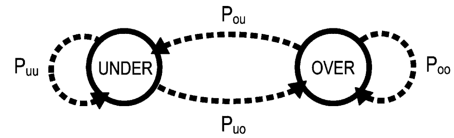

FIGURE 2. Diagram of transition probabilities between the compliance states of being either below a regulatory standard (UNDER) or above a standard (OVER) in a one-step stationary Markov chain process.

$$NB_{\mathrm{no\ testing}} = P_{\mathrm{u}}(\bar{B}_{\mathrm{outcome\ A}}) + (1 - P_{\mathrm{u}})(\bar{B}_{\mathrm{outcome\ B}}) \quad (3)$$

where $\bar{B}$ is the typical per day benefit of the specified outcome. In effect, "typical" net benefits of each outcome are weighed by their probability of occurrence to arrive at an overall expected net benefit per day where no policy is in place. Also, the net benefits of no testing are clearly the same under either FIB variability scenario. In calculating $\bar{B}$, we use a mean for the number of visitors per day $\bar{N}$ in eq 2. Since FIB levels are not actually measured in the no testing case, a representative level must also be chosen for the states of being under or over the standard. We will refer to these values as E_{u} and E_{o}, respectively. We recommend selecting representative FIB levels from the observed distribution at the case study site (in the case study, we chose the 20th and 80th quantiles, setting $E_{\mathrm{u}} = 50$ and $E_{\mathrm{o}} = 300$). We later show that the choice of representative FIB levels did not significantly influence the overall benefit comparison.

The testing policy is assumed to consist of daily monitoring with closures implemented for the 24-h period following detection of an exceedance, at a per day cost of C (e.g., the prorated cost of capital investment, labor, and sample processing). In the non-informative scenario, the joint probability of each day's management outcome is like the result of two sequential random draws from the historic distribution. The expected per day net benefits of testing under the non-informative scenario are therefore estimated as

$$\begin{aligned} NB_{\mathrm{non-informative\ testing}} = -C + P_{\mathrm{u}}^{2}(\bar{B}_{\mathrm{outcome\ A}}) + \\ P_{\mathrm{u}}(1 - P_{\mathrm{u}})(\bar{B}_{\mathrm{outcome\ B}}) + P_{\mathrm{u}}(1 - P_{\mathrm{u}})(\bar{B}_{\mathrm{outcome\ C}}) + \\ (1 - P_{\mathrm{u}})^{2}(\bar{B}_{\mathrm{outcome\ D}}) \quad (4) \end{aligned}$$

In the informative scenario, this equation is modified to reflect correlation in FIB values across daily time steps. We propose that the distribution of days when FIB was above and below a regulatory standard can be modeled as a discrete time Markov chain that has stationary transition probabilities across a two-state space and time steps $T = (0, 1, 2, ...)$, following Karlin and Taylor (*28*). With each time step, a transition occurs whereby the swim area either stays in the same state or changes to the other state. Let P_{yz} designate the one-step transitional probability that state z will occur at time step $T = n + 1$ given that state y occurred at $T = n$, where y and z each take on one of the two possible states, yielding four possible transitions. Figure 2 shows the four interrelated transitions in a two-state Markov chain. For the case study, we find no evidence to suggest that the mean and variance of *E. coli* levels are changing over the 4-yr time period under consideration, so stationarity is a reasonable assumption.

To calculate the transition probabilities, historic day to day transitions between FIB compliance states are analyzed with respect to an actual or hypothetical standard. In reality, many jurisdictions do not collect daily water quality observations. For instance, at our case study beach, *E. coli* data were collected only on Thursdays and on days following exceedances. This allows us to directly observe only two types of transitions. Fortunately, we can use unbiased estimates of P_u, P_{ou}, and P_{oo} to compute the two unobserved transitional probabilities (P_{uu} and P_{uo}) by solving the following balance equations:

$$P_u = \frac{P_{ou}}{P_{ou} + P_{uo}} \tag{5}$$

and

$$P_{uu} = 1 - P_{uo} \tag{6}$$

The joint probability of each management outcome is then obtained by multiplying the transitional probability by the background probability of being in the given state in any time step. The per day expected net benefits of testing under the informative scenario are thus:

$$NB_{\text{informative testing}} = -C + P_u P_{uu}(\bar{B}_{\text{outcome A}}) + P_u P_{uo}(\bar{B}_{\text{outcome B}}) + (1 - P_u)P_{ou}(\bar{B}_{\text{outcome C}}) + (1 - P_u)P_{oo}(\bar{B}_{\text{outcome D}}) \tag{7}$$

Equation 5 weights the net benefit values for typical days of each of the four management outcomes by their joint probabilities of occurrence. P_u and the transitional probabilities will clearly differ by location and perhaps over times of year, depending on local FIB conditions and the FIB standard and closure policy in place. This manner of representing FIB variability as a series of transitions between daily compliance states provides a high degree of flexibility to consider policy impacts at locations with differing FIB characteristics and for testing the sensitivity of conclusions to uncertainties about or changes in a local FIB system.

We formulate these equations as per day metrics to facilitate comparing policies either on the basis of single-day expectations or sums across whole seasons with policies that do testing at different intervals. Although we do not compute this value for our case study, the total season net benefits of a policy can be computed by weighting the per day net benefits by the total number of days in a season on which that policy was employed. For instance, the total season net benefits of a once-per-week testing program would be computed by adding 6 days of $NB_{\text{no testing}}$ and 1 day of NB_{testing} for each week the program is in place.

Case Study Application and Results

In this section, we assess the economic and epidemiologic implications of an idealized testing and swim closure policy using data from four recent summers at IDSP, a heavily used recreation destination on the southern shore of Lake Michigan about 50 km southeast of Chicago. We first describe the management setting and summarize our input parameters. We next use the best available local water quality data from four recent summers (1998−2001) to estimate a historical distribution of management outcomes and apply the transfer values identified previously to estimate the economic effects of typical open and closed days. We then compare the net benefits of no testing versus testing and explore the sensitivity of the analyses to the two different scenarios of *E. coli* variability and the values that visitors ascribe to health and recreation.

Background and Setting. Annual visitation to IDSP is about 750 000 persons. The swim season is generally limited to May−September; we will hereafter define this 152-day period as the summer season. Statistics on the number of cars entering the park in 1998 and 1999 and the average number of passengers per car (3.1) suggest an average summer daily visitation of 3240 people (*29*). Since visits are not evenly distributed during the week (a sunny summer weekend day or holiday may have up to 15 000 visitors), this may overestimate typical Thursday visitor numbers. However, it is the best available estimate. We assume that one-quarter of all visitors to IDSP swim, based on the results of a 1984−1985 study of visitor behavior at nearby West Beach in the Indiana Dunes National Lakeshore system (*30*). The per day cost of the program was estimated at $250 by a scientist with local sampling experience (*31*). Table 1 summarizes the transfer estimate parameters.

TABLE 1. Transfer Parameters for Indiana Dunes State Park Policy Analysis

parameter	symbol	transfer estimate	
swim closure management indicator	S	1 if swim area is open and −1 if swim area is closed	
fraction of visitors that swim or would swim	F	0.25	
no. of visitors per day	N	3,240	
value per visitor day of swim recreation (in 2000 U.S. dollars)	$V_{\text{recreation}}$	low	$16.02
		high	$38.46
value per visitor day of avoided health effects (in 2000 dollars)	V_{health}	mild	$280
		moderate	$1125
E. coli cfu/100 mL	E_u	under standard	50
	E_o	over standard	300
testing program cost per day	C	$250	

Weekly monitoring since 1991 has shown that summertime water quality conditions sporadically exceed the U.S. EPA's current 235 *E. coli* cfu/100 mL standard (*10*). Despite long-term investigations of *E. coli* occurrence at IDSP, the bacteria source(s) remain unclear. Currently, the practice of IDSP managers has been to take two samples each Thursday morning, one on each side of where Burns Ditch flows into the lake. Burns Ditch is a manmade channel that conveys mostly nonpoint runoff from IDSP's roads and wetlands. If on the following morning, the *E. coli* cfu density from either or both samples is found to be over the standard, we assume that the swim area is closed for 1 day. The swim area remains closed and sampling is continued daily until a future sample count is found to be under the standard. During the study period, swim closures averaged over five per year and most lasted only 1 day.

In analyzing the data for IDSP, we assume that the policy was uniformly and strictly implemented. Historically, management decisions sometimes deviate from stated protocols because of weather, expert judgment, or other factors. For instance, some closures are imposed for reasons other than high *E. coli* counts on the previous day. This produces a policy assessment that is representative of common FIB sampling and closure practices. Although we use actual visitor and water quality observations in our analysis, we are in no way representing that the outcomes described here are what actually occurred at IDSP. Our results demonstrate what would have occurred, given the observed and assumed conditions we present for IDSP, if the described policy was strictly applied.

Since IDSP did testing once per week except for repeat testing on days following exceedances, we possess information on *E. coli* levels complete enough to assess compliance status and management outcomes for only 118 of the 608 summer days. We restrict our policy assessment to these "managed days" only. We found no evidence that mean FIB levels on Thursdays varied significantly from levels on other

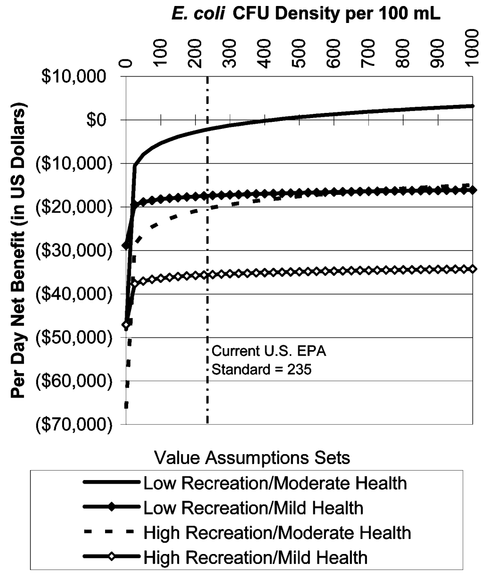

FIGURE 3. Typical per day net benefit of the Indiana Dunes State Park swim area when closed, by *E. coli* density under four value assumption sets.

TABLE 2. Typical Per Day Net Benefits for Swimming at IDSP by Management Outcome and Value Assumption Set

value assumption set (recreation/ health effects)	net benefits by management outcome: swim area open, A $E_u = 50$	swim area open, B $E_o = 300$	swim area closed, C $E_u = 50$	swim area closed, D $E_o = 300$
low/moderate	\$7 935	\$1 274	−\$7 935	−\$1 274
low/mild	\$18 859	\$17 201	−\$18 859	−\$17 201
high/moderate	\$26 106	\$19 444	−\$26 106	−\$19 444
high/mild	\$37 030	\$35 372	−\$37 030	−\$35 372

days, justifying our use of the set of Thursday observations ($N = 96$) as an independent, unbiased sample for use in estimating P_u. Repeat sampling days ($N = 22$) were partitioned out and used only in calculating the observed transitional probabilities P_{ou} and P_{oo}.

Typical Per Day Net Benefits When Swim Area Is Open or Closed. Figure 3 shows the estimated benefits for all swimming visitors for *E. coli* levels between 10 and 1000 when the IDSP swim area is closed, using eq 2 under all four value assumption sets. The benefit levels change rapidly at low *E. coli* levels because of the log function in the dose−response relationship used (eq 1). At *E. coli* levels above 50, however, the relative value assumptions begin to dominate the overall benefit level while increasing indicator density has relatively little influence. The upper bound benefits are defined by the low recreation/moderate health assumption set, and the lower bound is defined by the high recreation/mild health assumption set. An important finding is that swim closures only provide a positive net benefit to swimmers under the low recreation/moderate health assumption set and only at *E. coli* levels above 423 cfu (where the net benefits line intercepts the *x*-axis), well above the current recommended 235 standard. A graph depicting the net benefit to swimmers of an open swim area would look like a mirror image or reflection of Figure 3 about the *x*-axis.

Table 2 shows the typical per day benefits by management outcome and value assumption set at representative *E. coli* levels for each compliance state. Depending on the assumption set used, the benefits to the swimming population of a typical open swim day at IDSP beach when *E. coli* is under the standard ($\bar{B}_{\text{outcome A}}$) range between \$7935 and \$37 030.

TABLE 3. Historical Management Outcomes and Probabilities at IDSP, 1998−2001

year	management outcomes: A	B	C	D	background and transition probabilities: P_u	P_{uu}	P_{uo}	P_{ou}	P_{oo}
1998	28	5	2	0	0.848	0.821	0.179	1.0	0.0
1999	17	6	4	5	0.739	0.843	0.157	0.444	0.556
2000	17	5	5	1	0.773	0.755	0.245	0.833	0.167
2001	14	4	3	2	0.778	0.829	0.171	0.60	0.40
overall	76	20	14	8	0.792	0.833	0.167	0.636	0.364

The benefits of a typical open swim day when *E. coli* is modestly above the standard ($\bar{B}_{\text{outcome B}}$) are slightly lower but consistently positive, ranging between \$1274 and \$35 372. The wide dollar range across value assumption sets shows the sensitivity of the estimates to the relative magnitudes of the health and recreation values used.

Closures appear to cause a net economic loss of up to \$35 000/day for IDSP visitors that want to swim, regardless of whether the closure was a type I error ($\bar{B}_{\text{outcome C}}$) or justified by the policy ($\bar{B}_{\text{outcome D}}$). Lost recreation benefits to a large number of people on balance appear to outweigh the value of avoiding a small number of mild to moderate cases of gastrointestinal illness. Type II closure errors (outcome B) appear to do very modest economic harm in comparison, primarily because the number of persons expected to contract illnesses by permitting swimming on a day when *E. coli* = 300 (9 persons) is only slightly more than on a day when *E. coli* = 50 (3 persons). Although technically classified as an "error", a typical outcome B day produces highly *positive* benefits that are just \$1658−\$6661 lower than those of a typical outcome A day. Outcome D days, despite being an intended management outcome, produce negative benefits to swimmers under all four assumption sets. Assuming that managers control only the closure management choice and not the *E. coli* levels at the beach, it follows that closure errors can only be eliminated by shifting from outcome B to outcome D and from outcome C to outcome A, respectively (vertical movement in Figure 1). However, our results show that moving from outcome B to outcome D changes the net benefits from positive to negative. Avoiding unnecessary closures is therefore the most effective way, all else remaining equal, to increase net benefits to the swimming population.

Management Outcome Probabilities. Table 3 shows the distribution of management outcomes and the transitional probabilities of each outcome based on the 118 managed summer days at IDSP. Among the 96 independently sampled Thursdays, 76 days were under the standard (outcome A) and 20 were over it (outcome B). Thus, our estimate of P_u is 0.792. For comparison, comprehensive studies with much larger sample sizes taken at 63rd Street Beach in Chicago, IL (*8*), found P_u to be 0.95. Management outcomes from the set of repeat sampling days are used to calculate first the observed transitional probabilities P_{ou} and P_{oo}, and then the two unobserved probabilities are computed using eqs 5 and 6.

The data show that closure errors occurred under the current policy on about 29% of monitored days. There were a total of 28 regulatory exceedances. Owing to lack of timely water quality information, the swim area was left open on 20 days when the policy dictated it should have been closed. Of the 22 days when the swim area was closed, 14 closures (64%) were unnecessary and 8 (36%) produced the desired policy result. Combining the occurrence rates with the benefit estimates, we see that type I closure error days may have caused economic losses to IDSP swimming visitors of between \$111 088 and \$518 415 over the 4-year period, depending on the value assumption set used. Thus, up to

TABLE 4. Joint Probabilities of Management Outcomes for IDSP, 1998−2001

	joint probabilities by management outcome			
	A	B	C	D
year	$P_u(P_{uu})$	$P_u(P_{uo})$	$(1 - P_u)P_{ou}$	$(1 - P_u)P_{oo}$
1998	0.697	0.152	0.152	0.000
1999	0.623	0.116	0.116	0.145
2000	0.583	0.189	0.189	0.038
2001	0.644	0.133	0.133	0.089
overall	0.659	0.133	0.133	0.076

TABLE 5. Expected Net Benefits of No Testing vs Testing under Non-Informative and Informative *E. coli* Variability Scenarios for a Typical Day under Each of the Four Value Assumption Sets

	net benefits by policy		
	no	testing	
value assumption set (recreation/health)	testing	non-informative	informative
low/moderate	$6 547	$3 569	$4 000
low/mild	$18 513	$10 550	$10 657
high/moderate	$24 718	$14 169	$14 600
high/mild	$36 684	$21 149	$21 256

half a million dollars in additional recreational value could have been had if those unnecessary closures had been avoided.

Over a larger number of days, the joint probabilities shown in Table 4 are the expected distribution of management outcomes given the *E. coli* conditions at IDSP. Interestingly, the Markov chain balance equations compel the proportion of each error type (outcomes B and C) to be equal. In the net benefits equations to follow, this fact will mean that the B and C outcome benefits tend to cancel each other out, with the variation in *E. coli* level being the only parameter causing them to differ and only slightly at that. The historical management outcome rates at IDSP fit this prediction approximately although not perfectly, possibly because of small sample size.

Policy Comparison: Per Day Expected Net Benefits. We use the typical day value estimates from Table 2 and the probabilities from Table 4 as inputs to eqs 3, 4, and 7 to arrive at estimates of the typical per day net benefits that can be expected from a policy of no testing versus a testing policy under both the non-informative and the informative *E. coli* variability scenarios. Table 5 presents the results by value assumption set. The net benefits of testing robustly fail to exceed those of no testing. The informative and non-informative scenarios do not appear to influence the conclusion, perhaps because at IDSP the differences in the joint probabilities under each scenario are so slight.

We test the sensitivity of our results to changes in the probabilities of compliance by varying two numbers in the transition probability matrix while holding the other two constant. This permits evaluation of policies under hypothetical improvements or declines in overall water quality conditions. Specifically, we fixed P_{uo} and P_{uu} at their current IDSP levels (0.167 and 0.833, respectively) and allowed P_{ou}, P_{oo}, and P_u to adjust accordingly. The Markov chain balance equations ensure that P_u and the transitional probabilities remain a feasible combination and that the sum of the joint probabilities equals one. Figure 4a,b shows the expected net benefits per day of the policies over all feasible values of P_u from 0 to 0.85 under two value assumptions sets. There are ranges of high P_u levels under all value assumption sets where testing produces positive net benefits, for instance in Figure 4b when *E. coli* exceeds the standard on about 45% or more

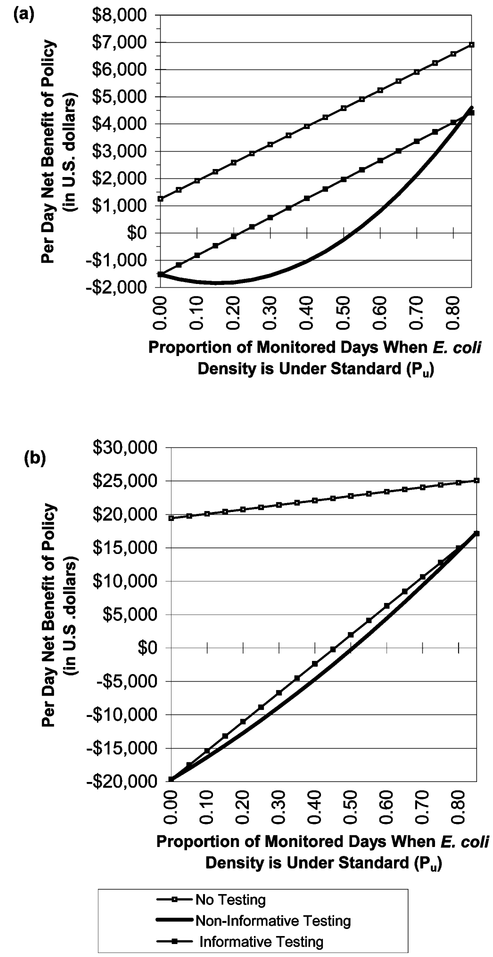

FIGURE 4. Per day expected net benefits of no testing vs testing swim closure policies under non-informative and informative *E. coli* variability conditions with P_{uo} fixed at 0.167 (a) under the low recreation/moderate health assumption set and (b) under the low recreation/mild health assumption set.

of monitored days. However, at IDSP, the expected net benefits of testing fail to exceed those of no testing across all theoretical levels of P_u. These findings invite comparison to conditions at other beaches.

As expected, the results of the analyses are sensitive to the health and recreation value inputs. Using a modestly higher WTP value for avoiding a moderate health episode ($V_{health} = \$1300$ instead of \$1125) causes closures to become net beneficial under the low recreation value assumption. Figure 5 shows that this difference also renders the policy comparison more ambiguous, by causing the informative testing scenario to become more cost beneficial than no testing at P_u values below about 0.62. In other words, under this hypothetical value assumption set, the benefits of testing begin to exceed those of no testing when *E. coli* exceeds the standard just slightly more often than it historically does at IDSP (i.e., on one-third or more of days). Thus, whether testing is economically justified may critically depend on the relative values that swimmers place on recreational access versus avoiding negative health effects and on the severity of health effects likely to be caused by locally occurring pathogens.

Policy Comparison: Predicted Cases of Illness and Lost Swim Days. The combination of eq 1 with the outcome probabilities in Table 4 suggests that testing and closure at IDSP may also prevent relatively few illnesses. Because our typical E_u and E_o days result in only 3−9 predicted cases of illness, respectively, a no testing policy at IDSP with $P_u = 0.792$ would result in 120 outcome A days with 411 total

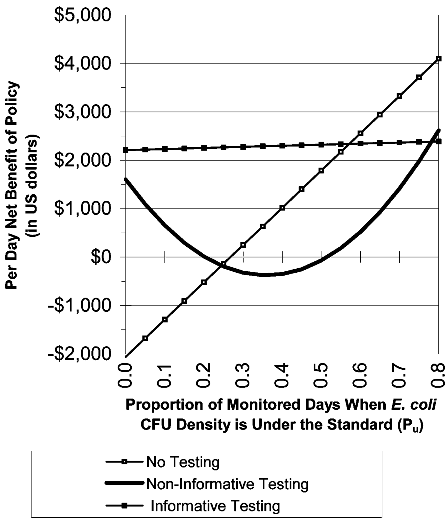

FIGURE 5. Per day expected net benefits of no testing vs testing swim closure policies under non-informative and informative *E. coli* variability conditions with P_{uo} fixed at 0.167 under the low recreation value assumption with the per visitor day health effect value set at $1300.

illnesses and 32 outcome B days with 299 illnesses. A closure policy, employed daily for the entire season, would prevent only about 178 illnesses or 22% of total predicted cases while causing an estimated 25 920 foregone visitor swim days. Even if the daily testing and closure policy could be implemented perfectly without errors, only 42% of predicted illnesses resulting from managed days would be avoided. This suggests that the majority of swim-related illnesses at IDSP may occur on days when swimming is permitted and *E. coli* levels are below the standard. This is consistent with a California marine beach simulation that estimated that 99% of illnesses occur on days when the swim area is open and in compliance (*32*).

It is critical to note that these estimates do not reflect any actual epidemiological data or records of illnesses attributed to water contamination at IDSP. The authors know of no evidence that any person has ever become ill from swim exposure at IDSP. Plausible reasons for this discrepancy include lack of diagnosis or reporting, but it is also possible that *E. coli* is a poor indicator for local pathogen conditions. Still, the transfer estimates presented here provide reasonable and useful approximations for assessing the plausible impacts of a policy and promote awareness of the fact that swim closure policies are at present neither designed nor able to eliminate most swim-related illnesses.

Discussion

Beach managers need realistic, cost-effective ways to examine their local situation and assess the relative effectiveness of current and proposed recreational water quality policies in relation to meaningful performance criteria. The absolute number of closures per season is an inadequate measure of the true local level of health risk or of the amount of illness prevention being achieved. The framework we have described provides a more comprehensive evaluation methodology. It is detailed enough to be applied at beaches with varying monitoring, ecological, and visitor circumstances but general enough to support broad policy development and incorporation of new or better information as it becomes available. The approach compels the scientist, manager, and analyst to clearly define the evaluation criteria and limiting assumptions, leading to identification of key information gaps. It is also adaptable to other environmental monitoring and policy issues involving risk indicators and dichotomous standards and decision-making.

We suggest that two important measures of the effectiveness of a recreational water quality policy are the number of predicted illnesses the policy is able to prevent and whether the chosen protocols generate closures with, at least on average, positive net economic benefits to swimmers. Our case study shows that commonly used swim closure practices perform poorly on both criteria under the *E. coli* and visitor circumstances at IDSP and may also perform poorly at much more polluted *and* cleaner swim areas. Swim closures—both those consistent with the policy and those in error—may not prevent very many illnesses and typically cause a net economic harm rather than benefit to the swimming population. These findings suggest that the water quality standard may be set at an inefficiently low level relative to dollar values that local swimmers ascribe to recreation versus health protection. An efficient standard would be set at the FIB density where, on average, the gains from closing the swim area exceed the recreation value lost. Closures at IDSP do not produce a positive expected net benefit until at least 423 *E. coli* cfu/100 mL and perhaps never. We also found that type I closure errors are much less harmful economically than type II errors while only modestly affecting illness rates relative to the open swim area status quo. At IDSP, net benefits to the swimming population may be greatly increased if the swim area were never closed at all.

A major purpose of this paper is to apply the rational lens of economics to shed light on what is at base an emotional issue. However, we have only presented a risk-neutral perspective. In reality, dollars probably are not nor should be the only decision factor in policy-making regarding risks. Individuals, and as a consequence our public decision-making institutions, tend to perceive financial gains and losses differently and disproportionately from their true dollar impacts. There are no "correct" answers as to what weights society ought to place on the trade-offs involved in any particular risk issue; they are context-specific and depend on society's preferences about the risks and alternatives being weighed. As examples, criminal justice systems often presume innocence at the risk of many guilty parties' going free, and drugs with severe side effects (including death) may receive approval if they offer large potential benefits (as in the context of patients with advanced cancers). The weights that society uses or would prefer to use regarding swim closure management outcomes have not been openly discussed in any literature of which we are aware and are a key area needing further research.

The ability of the framework to provide specific policy recommendations to beach managers will be improved as more precise, situation-specific values and relationships become available in four key areas: epidemiology, indicator variability, economic values for recreation and health, and community risk preferences. Sound epidemiologic studies are the obvious foundation upon which water quality standards and swim closure policies must be based. However, spatiotemporal variability and patchiness in the natural occurrence of FIB as well as uncertainty about the local association between FIB and various pathogens are significant obstacles for both epidemiologic studies and the development of meaningful indicator monitoring and closure protocols. Together, further epidemiologic and FIB ecology research are key to improving identification of public health risks and design of more effective policies.

Similarly, accurate value estimates and assessment of issue- and community-specific risk preferences are critical to the setting and evaluation of swim closure protocols. Values are necessary for comparing the cost-effectiveness and performance of monitoring or management alternatives. The absence of appropriate risk weights for swim closure policy-making is at present a significant barrier to being able to use value estimates directly in the process of setting standards and closure procedures. Behavioral and survey information about the risk preferences of actual and potential swimmers at specific sites would improve our ability to align local standards and management strategies with how local beach users actually regard the outcomes that are likely to occur. More precise information about all these factors, combined with increased public education and awareness, will enable a fuller and more grounded public debate about how cautious or lenient a system is desired for FIB monitoring and swim area management.

Although total elimination of closure errors may be unattainable, forecasting or faster detection of high FIB events could reduce error occurrence by enabling more accurate ex ante or near real-time closure decisions. There are several examples of both rapid testing technologies and FIB forecasting models (*33*, *34*) in development, but at present these alternatives are experimental and prohibitively expensive so their use in closure decision-making remains limited. Concerns also exist about the practical implications of using model-based forecasts or near real-time test outputs. Managers can use our framework to quantitatively and objectively test new policy or technologic innovations against baseline performance measures. Until alternative strategies and technologies become more accessible and scientifically and politically accepted, swim closure decisions may continue to be made using inadequate water quality information, possibly resulting in significant loss of recreation access with little public health gain.

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