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An Integrative Approach to Managing Coastal Bacterial Pollution

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Donald Bren School of Environmental Science and Management

by

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June, 1999

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ABSTRACT

An Integrated Approach to Coastal Bacterial Pollution

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Santa Barbara County has been testing ocean water for the presence of indicator organisms since 1995. Indicator organisms are bacteria used to indicate the presence of human waste and include total coliform, fecal coliform, and enterococcus. The levels of these organisms often exceed acceptable standards, resulting in beach closures. We studied the causes and impacts of bacterial contamination at two local beaches: Arroyo Burro and Arroyo Quemada. We took an integrated approach to the problem, examining areas of hydrology, microbiology, law, and economics. Watershed runoff from various land uses contributes to coastal bacterial contamination. We examined the hydrology of the watersheds affecting the two sites and found that Arroyo Burro, an urban watershed, produced greater runoff than Arroyo Quemado, a rural watershed, illustrating the effect of development on runoff. Microbiological analysis demonstrated the effects of precipitation, sediment suspension, and fertilizer on indicator levels. Levels of total coliform at ocean sampling sites were positively correlated with levels of precipitation and total coliform at a creek site increased after the addition of fertilizer to the watershed. Samples with sediment contained higher levels of total coliform than samples without sediment. When these sediments are suspended during a storm event, indicator organism loading to coastal waters can increase. Risk analysis predicted that illness rates for Arroyo Burro beach were higher during wet seasonal periods than in dry seasonal periods. Predicted illness rates were higher for Arroyo Quemada than Arroyo Burro year round. Legal analysis showed that the development of total maximum daily loads (TMDLs) for Santa Barbara County would be the most effective enforcement tool to reduce bacterial pollution. Best management practices (BMPs) required by NPDES Phase II regulations are difficult to enforce, but can be implemented voluntarily in conjunction with TMDLs to reduce bacterial loading within watersheds and thus beach pollution. We recommend delivery reduction BMPs that reduce sediment as well as bacterial loading; however, long term use of such treatment devices could adversely impact natural systems. Long term solutions require further research in the areas of DNA sampling, pathogens in lagoons, and land use effects on bacterial loading.

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Appendices

1.0   Introduction

Urbanization of coastal regions in the United States has been ongoing for over a century. More than 37% of the total United States population lived in urbanized coastal counties in 1990, and there are trends showing higher urbanization rates in coastal regions than in inland areas (NRC 1993). Coastal near-shore waters have been directly affected due to increases in urbanized population. The evidence suggests that throughout the United States, human related impacts on ecosystems are continuing at a rate that threatens the long-term sustainability of goods, services, and biological integrity provided by undisturbed coastal regions (NRC 1993).

Of particular interest is the attenuation of pollution loading into coastal waters from point and nonpoint pollution sources as a result of sewage spills and stormwater runoff coursing through areas where land use is comprised of a mix of urban, agricultural, and animal grazing lands. As stormwater moves through mixed land use watersheds, it acquires a variety of pollutants, including bacterial and viral pathogens originating in human or animal feces. Bacteria may be discharged from point-source locations such sewage treatment plants, leachate from landfills, and industrial sites. Moreover, bacteria may be picked up by stormwater as it moves through nonpoint source areas containing agricultural fertilizers, cattle fecal matter, septic system effluents, and urban animal and waterfowl feces (Howell *et al.* 1995).

Elevated concentrations of pathogenic bacteria and viruses have lead to widespread beach contamination and growing human health concerns. Coastal bacterial pollution has been shown to cause significant health and economic effects in waters throughout the United States. Studies have shown that this particular water pollution problem can pose swimming-related health risks (Cabelli *et al.* 1982; Fleisher *et al.* 1993) and widespread shellfish bed contamination (Legnani *et al.* 1998). As coastal urbanization continues in most coastal states, regulatory measures are being taken to reduce the impacts of nonpoint source pollution. Stricter enforcement of the Clean Water Act and the application of other coastal pollution abatement laws such as the Coastal Zone Management Act, aim to lessen pollution impacts in coastal regions of the United States.

Locally, Santa Barbara County has been experiencing problems with bacterial contamination along its beautiful coastline. In 1995, after realizing that ocean water at many of its beaches contained high levels of bacteria, the County of Santa Barbara began testing ocean water at eighteen different beaches for the presence of indicator organisms. Indicator organisms are bacteria used to indicate the presence of human waste in water. Testing for total coliform, fecal coliform, and enterococcus has revealed that on numerous occasions, bacteria levels have been above acceptable standards set by the USEPA (1986). When indicator bacteria are at unacceptable levels, the County either closes the beach or advises against contact with the water. The discovery of consistently high levels of indicator organisms has led to a

tremendous amount of public outcry. This public outcry has increased pressure on the County, and in response the County has undertaken a series of projects aimed at dealing with the pollution problem.

1.1   Santa Barbara County Projects

The description of the following projects was obtained from the Water Quality Program Report prepared for the Santa Barbara Board of Supervisors and the City Council.

In addition to the County's ocean monitoring plan, the Santa Barbara County Board of Supervisors, the Santa Barbara City Council, and the City of Carpinteria have undertaken three related water quality projects: Project Clean Water, the South Coast Watershed Characterization Study, and Project National Pollution Discharge Elimination System (NPDES). These three projects were initiated independently, but often overlap with respect to scope and agency cooperation. The projects will eventually be combined into one comprehensive water quality program.

Project Clean Water
Project Clean Water is a short-term study of seven watersheds throughout Santa Barbara County, including the Arroyo Burro and Arroyo Quemado watersheds. The project was initiated to address public concern about beach closures due to bacterial contamination during summer months when stream flow is low. The project includes creek walks to identify possible bacteria sources during low flow conditions, sampling for total coliform, fecal coliform, and enterococcus, public education and outreach, enhancement of enforcement programs, and the development of short and long term solutions to bacterial contamination.

The sampling portion of this project has been completed. Arroyo Burro creek was sampled four times a week from October 21, 1998 through November 17, 1998, and again in early December. Samples were taken at the creek outfall and at 13 points up the creek. Arroyo Quemado was sampled from October 26 to November 17, 1998. Samples were taken at the creek outfall and at 7 points up the creek.

The South Coast Watershed Characterization Study
The South Coast Watershed Characterization Study was designed to study the water quality of four watersheds in Santa Barbara County, including the Arroyo Burro watershed, over the 1998-99 storm season. The study was initiated to prepare for the future application of a NPDES permit by developing technical capability and sampling strategy. Four sampling events were scheduled to reflect baseline dry weather, first flush, middle season, and end of season conditions. A first flush event is specified as one that produces at least ¼ inch of rain in a 24-hour period and generates runoff.

Project NPDES

Santa Barbara County must apply for a NPDES Phase II permit by June 1, 2002. (United States Jan. 9, 1998) NPDES permitting is discussed further in the legal enforcement mechanisms portion of our report. Project NPDES was initiated to prepare for this permitting by developing a storm water runoff program. This program must incorporate public involvement, public education and outreach, an illicit connection and discharge detection program, a municipal operations control program, a construction site discharge control program, and a new development control program. The data collected by the South Coast Watershed Characterization Study and Project Clean Water will be used in developing and implementing Project NPDES.

1.2   Stakeholders

Throughout the Santa Barbara region, there exist concerned individuals and interest groups who have a vested interest in the contamination of Santa Barbara County's coastal waters. These stakeholders include:

- Community Members
- Citizens Groups
- Local Business
- Non-Profit Environmental Organizations
- City of Santa Barbara
- Santa Barbara County Environmental Health & Safety
- Santa Barbara Public Works and Utilities
- California Regional Water Quality Control Board (RWQCB)
- US EPA

Individuals and citizen groups concerned with the degradation of coastal water quality have expressed their concerns to local and regional governments. Local governmental agencies and non-governmental groups have responded by forming coalitions to address the pollution problem. Local businesses are also interested in a solution to the problem. If tourists and beach goers decide to avoid county beaches, local businesses could suffer. Finally, non-profit environmental groups such as the Surfrider Foundation, the Santa Barbara Community Environmental Council, and Heal the Ocean are also putting forth an effort to solve water quality problems.

1.3   Project Objectives

We believe that in order to develop effective solutions to complex environmental problems, such as the reduction of nonpoint source pollution in watersheds, a multidisciplinary approach is necessary. This thesis project has been constructed to

gain knowledge about various scientific, legal, and political factors inherent to the problem of and solutions to coastal bacterial pollution. We set out to determine how watershed functions affect levels of indicator organisms at the beach. Specifically, we hypothesized that precipitation, runoff, and sediment loading increased indicator bacterial loading. Although we acknowledged that increased levels of indicator organisms might not accurately predict increased levels of pathogens, we hypothesized that the risk of adverse human health effects increases with increased levels of indicator organisms at beaches. Finally, we attempted to determine what legal enforcement mechanisms and best management practices are available to reduce levels of indicator organisms at beaches.

With these hypotheses in mind, the following are five main objectives of our thesis project that include investigations into various scientific, legal, and environmental management areas:

1. Investigate the effect of storm water runoff on bacterial loading to coastal waters.
2. Establish the role of precipitation, sediment re-suspension, and nutrient loading in bacterial contamination.
3. Determine the human health risks associated with swimming at beaches polluted with bacteria.
4. Identify the legal mechanisms available to reduce coastal bacterial pollution.
5. Identify possible best management practices (BMPs) to reduce coastal bacterial pollution.

Each objective listed above is associated with a variety of complex physical and biological relationships. In addition to addressing the above objectives using models and experimentation, we substantiate our conclusions using the existing body of literature on bacterial transport processes, hydrological processes, health risk relationships, and current legislative frameworks.

Our project report is broken down into five sections to address these issues. The sections include hydrology, microbiology, risk analysis, legal enforcement mechanisms, and best management practices. We investigate how these five disciplines can be used together in a cohesive framework for watershed analysis. Results from each individual section are used in the other respective analyses to create an interdisciplinary approach to the study.

1.4   Study Sites

Many of the beaches sampled in the County's ocean monitoring program have shown transiently high indicator bacteria concentrations. However, the coastal waters adjacent to Arroyo Burro County Beach and Arroyo Quemada Beach have suffered from chronically high levels of fecal bacteria. Arroyo Burro beach was out of

compliance for at least one indicator organism for 17 out of 52 weeks in the 1997-98 year. Arroyo Quemada was out of compliance for at least one indicator organism for 41 out of 52 weeks during the same year. Arroyo Burro Beach (Hendry's Beach), located just outside Santa Barbara City limits, is a popular attraction for both local residents and tourists to the area. Arroyo Quemada Beach, located approximately twenty-three miles west of the City limits, is a small, semi-private beach used primarily by members of the adjacent Arroyo Quemada Lane community. Appendix I shows the location of both study sites.

The Coastal Bacteria Contamination Group Project studied the three watersheds that directly impact the beaches of the two study sites. These watersheds are:

- Pila Creek
- Arroyo Quemado Creek
- Arroyo Burro Creek

Both the Pila and Arroyo Quemado Watersheds discharge at Arroyo Quemada Beach. The Arroyo Burro Creek Watershed discharges at Arroyo Burro County Beach. These watersheds will be described in greater detail in the hydrology section of this paper.

2.0   Microbial Ecology

2.1   Background

2.1.1   Indicator Organisms

Microorganisms in water are known to cause diseases in human beings. In order to protect public health, it is imperative to detect the organisms that are pathogenic. However, it is not practical to sample directly for all pathogenic organisms because of the multifarious array of the pathogens in the environment. Therefore, indicator organisms are selected to demonstrate the presence of human waste and hence the potential presence of pathogens. Indicator organisms are usually of intestinal origin from endothermic animals. Therefore, the presence of these organisms in the water indicates contamination of fecal matter, which could also contain pathogens such as *Salmonella* and *Shigella*. EPA regulations require three main groups of indicator organisms to be used to monitor water quality; they are total coliform (TC), fecal coliform (FC), and enterococcus. The coliform group is defined as "all the aerobic and facultatively aerobic, gram-negative, non-spore-forming, rod-shaped bacteria that ferment lactose with gas formation within 48 hours at 35°C" (Madigan *et al.* 1997). Fecal coliforms are coliform bacteria that live in the intestines of warm-blooded animals. Enterococcus is the umbrella group for fecal streptococcus, usually present in feces of endothermic animals. Among the three indicators, enterococcus has been shown to have the highest correlation with gastroenteritis in marine waters (Cabelli 1982). Enterococci also survive longer in the environment than coliforms (Anderson

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et al. 1997). The assumptions made in using indicator organisms are that 1) the disease-generating bacteria die off sooner than TC, FC, and enterococci outside of human or animals bodies; and 2) both the indicators and pathogens are affected the same by water purification processes (Madigan *et al.* 1997).

Questions arise when using indicator organisms because there are the possibilities of "false positive" and "false negative" results. False positive results occur when the indicators are present in the water while the pathogens are not, leading to needless preventive measures. False negative results occur when the pathogens are present in the water while the indicators are not, leading to public health hazards. The reliability of the indicator organisms expressing the presence of hazardous bacteria and enteric viruses is the subject of much research and will be addressed further in the next section of this report.

Correlation of indicator organisms with pathogenic organisms

Studies have been published that show correlations between indicator organisms and pathogenic organisms. Polo *et al.* (1998), Van Donsel and Geldreich (1971) and Efstratious *et al.* (1998) found the levels of total coliform, fecal coliform, and enterococcus to be significantly higher in waters tested positive for *Salmonella* than in waters tested negative for *Salmonella*. This is an important result because *Salmonella* is a common cause of gastroenteritis in humans (Polo *et al.* 1998). TC, FC and enterococci also have moderate positive correlations with *Staphylococcus aureus* and *Candida albicans*. In the Efstratious *et al.* (1998) study, total coliform had the strongest correlation with *Salmonella* and *S. aureus*. The combination of total coliform and fecal coliform was best at predicting the occurrence of *C. albicans*.

Wyer *et al.* (1995a) studied the relationship between total coliform, fecal coliform, and fecal streptococcus and the presence of enterovirus in coastal waters. They found no significant statistical relationship between the concentration of fecal indicator species and the presence of enterovirus. In their study, the concentration of fecal indicator organisms in sewage effluent remained nearly constant while the numbers of pathogens varied widely. They found that the numbers of pathogens varied as the levels of disease varied in the local community.

These studies do not necessarily contradict each other about the benefits of indicator organisms since these researchers were studying different pathogens. It is important to note TC, FC, and enterococcus may be effective at predicting some pathogens but not all.

Problems with using enterococcus as an indicator

The study by Anderson *et al.* (1997) of enterococci in New Zealand waters found enterococcus to be a less than ideal measure of human pathogen presence. They found that enterococci were able to persist on degrading seaweed, live longer than coliforms,

and to be spread by numerous animals including insects. This finding demonstrates that the existence of enterococcus does not necessarily indicate the presence of fresh human fecal material. Furthermore, they noted that birds have high enterococci levels. Since enterococcus is found in all warm-blooded animals, and human pathogens are higher in human feces than non-human feces (Jagals and Grabow 1996), high enterococcus levels do not always indicate a large threat to human health. The fact that enterococci can survive and reproduce outside of the host indicates that high levels of enterococci do not always signify high levels of any fecal waste.

Developments in indicator detection

Since the use of planktonic total coliform, fecal coliform, and enterococcus as indicators of fecal pollution is relatively crude, many researchers are searching for other ways to detect the presence of pathogenic organisms. Improvements are being made both in sampling and in detection methods. Many proposed detection methods are molecular techniques, which can be much quicker than the current widely used enzyme assays. The methods we report on include quantitative PCR, which is a measure of the amount of different types of DNA present in a sample, use of *Bacteroides* probes, sorbitol fermenting bifidobacteria, "Microbial Source Tracking", and a sediment bag sampling method.

Fricker and Fricker (1994) studied the use of polymerase chain reaction (PCR) in detecting *E. coli* in water. They found that PCR efficiently amplified *E. coli* in water. This method detects *E. coli* in only three hours, which is significantly faster than the enzyme assays currently employed. Therefore, contaminated beaches can be closed in a timelier manner than with the enzyme assay method, which is explained in section 2.1.2. Two drawbacks to this method are that the primer sequences used to amplify *E. coli* also amplify non-*E. coli* coliforms as well as DNA from both live and dead bacteria.

Jagals and Grabow (1996) evaluated the use of sorbitol-fermenting bifidobacteria as indicators of human fecal pollution of water. These researchers used sorbitol-fermenting bifidobacteria because these bacteria are known to be specific to human feces. They found sorbitol-fermenting bifidobacteria to be a more reliable indicator of human waste than fecal coliform or fecal streptococci. One problem with using sorbitol-fermenting bifidobacteria as an indicator is that it is relatively short lived in the environment. However, certain environmental conditions allow these organisms to persist, which enhance their use as an indicator.

Kreader (1995) studied the use of *Bacteroides* DNA probes to detect human fecal pollution. In this study, PCR primers specific for the 16S rRNA sequences of *Bacteroides distagonis*, *B. thetaiotamicron*, and *B. vulgatus* were used. Of the human fecal extracts tested, 67 to 78% had high levels of target DNA, while only 7 to 11% of the non human fecal extracts contained high levels of target DNA. These findings suggest that the use of this probe may be useful in distinguishing fecal pollution from

human and non-human sources. However, *B. vulgatus* was detected in 63% of house pets, so this particular probe would not be useful in distinguishing pollution from house pets and sewage or septic tank leaks in suburban runoff. Like sorbitol-fermenting bifidobacteria, *Bacteroides* also persist for only a short time after entering natural waters, so further study is warranted.

In King County, Washington State, Samadpour (1995) has developed a method termed "Microbial Source Tracking" to determine the types of fecal pollution in water. First, water samples are collected and analyzed for fecal coliform. Isolates are obtained from each sample and analyzed to positively identify *E. coli*. DNA is then extracted from these isolates. Each DNA sample is digested with restriction endonuclease, then run on an agarose gel by electrophoresis. The DNA fragments are hybridized with a ribosomal RNA probe. The blots are exposed to X-ray film, producing an audioradiogram that shows the banding pattern (ribotype) of the DNA. Samadpour uses a database of *E. coli* strains and matches up the strains found in streams with the database in order to determine the animal origin of the fecal matter. Currently, the *E. coli* that can be distinguished includes human, goat, pig, horse, cat, dog, and llama. This method may be very useful for areas that contain unknown fecal pollution. It may also be useful in determining the relative pathogenicity of the fecal pollution in question to humans.

Nix *et al.* (1994) reported on a novel sampling strategy for mapping the plume of indicator bacteria in the environment using porous (sediment) bags filled with sand. The sediment bags are suspended with buoys in a grid pattern in the water where sampling is desired. Indicator bacteria such as fecal coliform will accumulate in the sandbags, thus retaining the bacteria until the sediment bags can be analyzed for bacterial counts. The pattern of the bacterial plume and the sources may be elucidated with the sediment bags.

2.1.2   Bacteria Enumeration Method

Statistical Basis of Bacteria Enumeration

Currently, enumeration of total coliform, fecal coliform and enterococcus by monitoring agencies is based on the multiple tube fermentation method, using the most-probable-number (MPN) as the statistical basis. The enumeration of indicator organisms by the MPN procedure is not a direct measurement of pathogenic microorganisms. The MPN procedure provides an indirect estimate of bacteria by inoculating serial dilutions of bacterial suspension in liquid culture and analyzing observed growth in each of the dilutions. The mathematical basis of the MPN is based on a Poisson probability model developed for bacterial quantification by interpreting fermentation tube results (Beliaeff and Mary 1993). Baliaeff and Mary (1993) calculated and compared the bias of using Poisson distribution and binomial distribution in estimating the MPN. Their result indicates that the binomial model yields less bias than the Poisson model.

Careful consideration is warranted when utilizing the MPN results to make policy decisions. It is difficult to conduct a direct count of bacteria in ocean water. Therefore, random sampling and statistical probability models are used in order to obtain a thorough representation of the bacterial level in the water. As a result, there are variances and sampling errors that compromise the ability of the laboratory results to represent accurately the bacterial density in the water. For an infinite number of samples, the MPN result is unbiased. However, when indicator bacteria are enumerated with the multiple tube fermentation method there is an under estimation of the true bacterial density by the MPN because of the small number of tubes used. Also, the number of dilutions must be adequate in order to get an accurate result because there is an upper bound for the MPN for each dilution. Therefore, if the sample is not diluted enough, the upper bound will be reached, and the true number of bacteria per 100 mL sample will be unknown. This may be adequate for closing beaches since the upper bound will be higher than the water quality standard, but this is inadequate if the true numbers are desired to conduct a statistical analysis.

Laboratory Analysis

The Santa Barbara County laboratory previously used the multiple tube fermentation method for bacteria count analysis. However, due to the time consuming nature of the method, techniques developed by IDEXX (Westbrook, ME, USA) are presently used. Pre-made packages brand named Colilert® and Enterolert®, as well as Quanti-Tray™ and a buffer solution distributed by Butterfield Inc. facilitate rapid analysis, minimize messiness and still utilize MPN as the statistical basis.

The Colilert® reagent is a defined substrate technology that simultaneously detects total coliform and *E. coli*. The Colilert® test uses the nutrients ortho-nitrophenyl-β-D-galacopyranoside (ONPG) and 4-methyl-umbelliferyl-β-D-glucuronide (MUG) to incubate the indicators. The enzyme β-galactosidase from the coliforms hydrolyzes ONPG and releases ortho-nitrophenyl, which causes a yellow coloring. *E. coli* produces an enzyme, glucuronidase, that hydrolyzes MUG into glucuronide and 4-methyl-umbelliferone. The latter fluoresces under 365 nm ultraviolet wavelength (Pepper *et al*. 1995). Thus if the sample solution is yellow in color and fluoresces under UV light, TC and *E. coli* are present in the sample. The Enterolert® test operates on the same theory as Colilert®. Enterococci produce β-glucosidase, which lyse the β-glucoside on the MUG molecule, and the 4-methyl-umbelliferone fluoresces. Therefore, if enterococci are present, the solution will fluoresce.

Quanti-Tray™ operates on the same principle as the 15-tube multiple tube fermentation where bacterial counts in MPN/100ml are obtained. The 100 mL of sample is poured into the Quanti-Tray™ and divided into the proper portions. The IDEXX Quanti-Tray™ method has a counting range of over 2,000 MPN and more accurate results than the 15-tube multiple tube fermentation method. In addition,

using Quanti-Tray™ for bacterial enumeration is easy, rapid and economical, which is ideal for processing large amount of samples (Fricker *et al.* 1997). Research by Shadix *et al.* (1993) has shown that "the nutrient agar plus MUG method is a reliable method for detecting *E. coli*." Other researchers have also shown that Colilert® and Enterolert® yield accurate bacterial counts (Budnick et al. 1996; Clark and El-Shaarawi 1993; Cowburn *et al.* 1994; Eckner 1998; Fricker and Fricker 1994; Fricker *et al.* 1997).

Despite the research performed on the accuracy of using the Colilert®-MUG reagent for TC, *E. coli* and enterococcus, Landre *et al.* (1998) found that a false-positive result from the Colilert® test can occur due to the presence of *Aeromonas* in the water sample when the Colilert® reagent is within 4 weeks of the shelf-life expiration date. A study by Davies *et al.* (1994) also found that plant extracts and algae could interfere with the detection of these indicators. Their conclusion is that masking agents need to be developed for testing waters that contain high plant or algal biomass. The false-positive result from the Colilert® reagent presents less of a public health hazard than a false-negative result. The consequence of an artificially high bacterial count is only the unnecessary beach closure.

2.1.3   Bacterial Loading From Sediment

The Enterolert® and Colilert® procedures that Santa Barbara County currently uses are sensitive to water samples containing sediment. If a water sample contains a large amount of sediment it is impossible to analyze the results of the Colilert® and Enterolert® tests because the yellow and fluorescent colorings do not show. Therefore, the county sometimes does not sample during rain events. This can be a problem if rain causes sediment re-suspension and loading of the bacteria into the water column. TC, FC and fecal streptococci are found to be 100 to 1000 times higher in the sediment than in the water column (Van Donsel and Geldreich 1971). Thus, during rain, a large number of bacteria are suspended in the water. Research by Davies *et al.* (1995) shows that fecal microorganisms are able to survive in the sediment because the solid particles provide a favorable, "non-starvation" environment for the bacteria. When the sediment is re-suspended by storm events or recreational activities such as wading and boating, bacteria are re-loaded into the water, thus creating a health hazard (Marino and Gannon, 1991). Therefore, a better monitoring method to analyze bacteria in the sediment must be administered. LaLiberte and Grimes (1982) contend that "the enumeration of sediment-bound fecal coliforms in high-use recreational areas and in food production waters (i.e., shellfish and irrigation waters) should be considered of equal importance to bacterial density determinations in the water column; the two are closely related." The county currently does not sample for the bacterial level in the beach sediment; thus, monitoring may not provide full protection of public health.

2.1.4   County Regulations

In accordance with California Assembly Bill 411, Santa Barbara County places a beach on advisory or closed status if it either exceeds a 5 sample geometric mean standard for recreational water, or if it exceeds the one time sampling standard for recreational water. For total coliform, the 5 sample geometric mean standard is 1000 MPN. The one time standard for total coliform is 10,000 MPN. For fecal coliform or *E. coli*, the five-sample standard is 200 MPN, and the one time standard is 400 MPN. The five-sample standard for *Enterococcus* is 35 MPN and the one time standard is 104 MPN. If one or two indicators exceed the standard, the beach is placed on advisory status. If all three indicators exceed the standard, the beach is closed (US EPA 1986).

2.2   Microbial ecology methods

One of our primary hypotheses was that sediments harbor indicator bacteria, which are then released into the water column during precipitation events. This would cause higher readings at ocean monitoring sites after rain. We also hypothesized that fertilizer application in the watershed may amplify the indicator organism concentrations by supplying inorganic nutrients. In order to test these hypotheses, we analyzed bacterial, fertilizer and precipitation data statistically from Arroyo Burro and Arroyo Quemada to find out if there was a correlation between precipitation, fertilizer and bacterial counts. We also conducted a sediment experiment that analyzed the difference in bacterial counts between samples with sediment and without sediment. In addition, we performed a literature search on the factors affecting bacterial levels in the environment.

2.2.1   Precipitation correlation

Precipitation correlation methods

In order to study our hypothesis that precipitation and associated runoff is positively correlated with higher indicator organisms in water, we investigated the correlation between precipitation and bacterial counts. Since there was no summer precipitation in Santa Barbara, it was not possible to test this correlation for year round data. We instead used the data beginning with the first rain event of the season for both years, and analyzed the data until the bacterial data had no associated precipitation events. The first method we employed was a straight linear regression. From the County of Santa Barbara, we obtained ocean monitoring bacterial data from Arroyo Burro and Arroyo Quemada (Oct. 1996-April 1998), and rain gauge data from the Arroyo Burro and Arroyo Quemado watersheds. Since bacteria were monitored on a weekly basis, we used the cumulative precipitation for the week prior to the sampling date without including precipitation from that date.

We graphed weekly cumulative precipitation and total coliform counts in beach water samples of Arroyo Burro and Arroyo Quemada. Figures 1 and 2 of Appendix II show that there is a clear correlation between the total coliform count and cumulative precipitation at specific time intervals. During high precipitation events (January 1998) the total coliform in the ocean is correspondingly high. The increase of total coliform without rain events is discussed further in a later section.

We log transformed the precipitation and ran a linear regression with bacteria data, and cube root transformed the precipitation and ran a linear regression with the bacteria data. We used these transformations of precipitation on the recommendation of Dr. Joel Michaelson, a climatologist in the geography department at the University of California, Santa Barbara. The highest correlation we found was an r^2 of .21 for the combined water years of 1996 and 1997 for Arroyo Burro, using a cube root transformation of precipitation. Since we found a weak correlation using these methods, we used Spearman's rank order correlation method, using a correction factor for tied ranks. Spearman's rank order correlation is a non-parametric correlation method that makes it possible to find correlations between variables that are correlated, but perhaps non-linearly (Ott 1993).

We ranked all data for each variable and applied the following correction formula for tied ranks:

$$R_t = \sqrt{A^2 + (n^2 - 1)/12}$$

(Equation 2.1)

R_t = corrected rank of the tied score
A = average rank of the tied score
n = the number of scores tied for one rank
(Senders 1958)

We then used the Spearman's correlation formula to find the correlation coefficient:

$$r_s = 1 - 6\Sigma D^2_I / N(N^2-1)$$ (Equation 2.2)

r_s = Spearman's correlation coefficient
ΣD^2_I = sum of squared differences between the two variables
N = number of cases in the distribution
(Senders 1958)

Appendix III contains the weekly monitoring data for the periods that we calculated the correlation. Table one contains Arroyo Quemada data and table two contains

Arroyo Burro data. Cumulative week precipitation equals the cumulative precipitation for the week prior to the sampling date without including precipitation from that date. The numbers under total coliform are the most probable numbers found by the county. Both the precipitation data and the total coliform data were ranked from low to high, 1,2,…,n. In the case of Arroyo Quemada, there were no ties so these ranks stood in the analysis. Arroyo Burro did contain tied ranks, so we used the correction formula stated above (Equation 2.1). The columns Rt, tc, and Rt precip contain the corrected ranks for total coliform and precipitation, respectively, that we used in our calculation. D is the difference between the precipitation rank and the total coliform rank.

Results of the statistical relation between precipitation and bacterial counts

Using the rank order correlation, we found a correlation coefficient of .75 for the combined precipitation years of 1996 and 1997 for Arroyo Burro. The r^2 was .56. We employed this same method for the 1997 precipitation year for Arroyo Quemada, but only found a .43 correlation coefficient. The corresponding r^2 for Arroyo Quemada was .21. The z statistic for the Arroyo Burro correlation was 5.29, with a corresponding p value of <0.00000029, which is significant at the < 0.001 level. Therefore, we reject the null hypothesis that precipitation and bacteria counts are not correlated. The z statistic for Arroyo Quemada was 1.52, with a p value of 0.0643. Therefore, the correlation for Arroyo Quemada is significant at the .128 level. We cannot reject the null hypothesis that there is no correlation between precipitation and bacteria counts for Arroyo Quemada.

Discussion of the precipitation and bacteria correlation

The high correlation between the amount of precipitation falling a week prior to the bacterial sampling date demonstrates that causation between rainfall and bacterial loading is likely. Although the correlation for Arroyo Quemada is not significant, it is important to keep in mind that only 1997-1998 bacterial data were available for Arroyo Quemada. Furthermore, unlike Arroyo Burro, Arroyo Quemada did not receive any precipitation between September 2, 1997 and November 10, 1997. Therefore, the statistical analysis did not begin for Arroyo Quemada until November 17, the first sampling date after the rain event, which occurred November 10.

Possible reasons for an increase in bacteria counts after precipitation events include re-suspension of bacteria from sediments, agricultural runoff, sewer line breaks, raw sewage releases, and general urban storm runoff.

2.2.2   Sediment Experiment

In order to further investigate our hypothesis that bacterial loading from sediment occurs during rain events, we performed a sediment experiment. In the fall of 1998, we sampled Pila Creek, located at the Santa Barbara County Tajiguas landfill, and

emptying onto Arroyo Quemada beach. We analyzed the samples using the 5-tube multiple tube fermentation method with Colilert-18®. The two types of water samples analyzed were samples with and without sediments. The main purpose of the experiment was to simulate bacterial loading into the water column during a rain event. In addition to elucidating the difference in bacteria counts of samples with and without sediment, we explored techniques for analyzing water samples with sediments and compared our bacterial count results with the county laboratory's results.

Materials and methods

We sampled Pila creek on four different dates (9/15, 9/22, 10/7, and 11/5). We took two grab samples each time with sterile two liter plastic bottles (product of Nalgene, Rochester, NY). The first sample was taken without disturbing the sediments. The second sample was taken after stirring the top two centimeters of sediment. We split the sediment-free sample into four replicates. We analyzed three of these replicates, and one was taken to the Santa Barbara County Public Health Laboratory for Santa Barbara County Solid Waste and Utility Division's creek monitoring project. We split the sample containing sediment into three replicates, each of which we analyzed. Approximately one liter was collected each time, and distributed into 100 mL sample collection bottles containing sodium thiosulfate (IDEXX, Westbrook, ME, USA). The two liter bottles were inverted to mix prior to pouring into each sampling bottle. An exception to this method occurred on September 22, 1998, when we instead stirred the top ten centimeters of sediment.

We enumerated total coliform and *E. coli* with Colilert-18® and 5-tube multiple tube fermentation method using dilutions of 10mL, 1 mL, 0.1 mL, and 0.01 mL. We made the dilution medium by adding nine Presence/Absence (P/A) Colilert® reagent packs to 900mL of sterile water from the Nanopure™ Ultra purification system (Barnstead, Dubuque, IA, USA). This was mixed until all reagents dissolved completely. One P/A Colilert® pack was added to each 100 mL water sample and shaken by hand vigorously for one minute. 10 mL of well-mixed sample was distributed to each of the 5 sterile test tubes using aseptic technique, and then diluted with the dilution media to 1 mL, 0.1 mL and 0.01 mL for each test tube. The field samples with sediments were analyzed with the same procedures as the samples without sediments. The sediments did not obscure sample results. We incubated the test tubes in a NewBrunswick Scientific incubator (Edison, NJ, USA) at 35°C for 18 hours. Total coliform was present if the water samples turned yellow. *E.coli* was present if the samples fluoresced under a 6 watt 356 nm UV lamp (FischerScientific). A Most Probable Number (MPN) out of 100mL was obtained from Table 9221:IV of *Standard Methods For the Examination of Water and Wastewater* (Clesceri *et al.* 1995).

We also explored alternative analysis procedures for enumerating bacterial count in muddy samples. We centrifuged samples with sediments in Nalgene plastic bottles with Sorvall® RC 5Bplus centrifuge (Newtown, CT, USA) at 15,180g for 10 minutes and extracted the supernatant without the sediment. We analyzed the supernatant

using the same procedures as samples without sediments. Another method for analyzing samples with sediments involves letting the particulates settle for 1 hour until the liquid is clear. When using the Quanti-Tray™ to enumerate bacterial counts, the sediment in the muddy samples settles, leaving the solutions in the wells clear. The results can be analyzed and the sediments will not obstruct interpretation of results if the sediments are not disturbed in the wells.

The means of the three replicates for each sample were obtained and used in the comparison. The resulting data were not distributed normally and contained outliers. However, since we had a small sample size we did not delete the outliers. Instead of using the Student t-test to compare the means, we used the Wilcoxon test (Ott 1993) to compare the two means at α=0.05. There are three assumptions for the student t-test:1) the two samples are independent, 2) the samples are drawn from Normal populations and 3) the variances from the two populations are equal (Ott 1993). Since our samples did not satisfy the assumptions for the student t-test, we used the Wilcoxon test. The only requirement for the Wilcoxon test is that independent random samples are taken from two populations.

Sediment Study Results

Please refer to Appendices IV and V for sampling results.

We obtained the results of the comparison of means of TC between the samples with sediments and without sediments through the Wilcoxon rank-sum test at the 95% confidence interval. We obtained a Z-score of 2.1781 and a p-value of 0.0147; therefore, the null hypothesis that the two means are the same was rejected, and the alternative hypothesis was inferred. The statistical results indicate that samples with sediment have higher TC counts than samples without sediments (See Figure 1, Appendix IV). The standard deviation of TC between samples with sediments and samples without sediments was compared using the one-sided Wilcoxon rank-sum test. The p-value obtained was 0.0571, indicating that we cannot reject the null hypothesis that the variances were equal. Therefore, the variability of the two types of samples was found to be the same.

We again used the Wilcoxon rank-sum test to compare the means of *E. coli* between samples with and without sediment. The p-value obtained from was 0.3429, indicating that we cannot reject the null hypothesis that the two means are the same. In addition, a two-sided Wilcoxon rank-sum test was also performed where the p-value found was 0.6250; therefore, the null hypothesis was still not rejected. The statistical analysis indicates that samples with sediment and without sediment have the same *E. coli* counts (See Figure 2, Appendix IV). The standard deviations for the *E. coli* counts for the samples with sediment were compared to samples without sediment, and the p-value was 0.3429, indicating that the variability among the samples is the same.

We compared the means of TC counts of the split samples analyzed by the Santa Barbara County laboratory and this project with a Wilcoxon signed-rank test. Each sample in this analysis was paired with another sample sent to the Santa Barbara County Laboratory, therefore, we needed an appropriate statistical test to compare paired data. We used the signed rank test because the distribution of differences between the county and our results contained outliers. The signed-rank test utilizes the sign and magnitude of the rank of the differences between pairs of measurements (Ott 1993). The p-value obtained was 0.1875, indicating that we cannot reject the null hypothesis that the two means are the same. Therefore, our results are equivalent to the County laboratory's results (See Figure 1, Appendix IV). The variance of our results was also the same statistically as the variance of the County's results (p-value = 0.6875).

We again used the Wilcoxon signed-rank test to compare the means of *E. coli* counts found by the county laboratory and by our laboratory. We obtained a p-value of 0.6875, indicating that the means were statistically the same. Therefore, our result is the same as the County's result (See Figure 2, Appendix IV). The variance of our results is also the same statistically as the variance of the County's results (p-value=0.8125).

Sediment Study Discussion

We split samples with the county in order to compare our results with the county's results. The county laboratory used the Colilert® reagent and Quanti-Tray™ quantification method, and we used the multiple tube fermentation and MPN table to obtain bacterial counts. Even though the data from 9/15 and 9/22 show that the county samples are higher than samples without sediment (See Appendix V), the county samples for 10/7 and 11/5 are lower and the same as our samples, respectively. The county data for 9/15 have a high standard error (mean = 2969, standard error = 1768); thus, the data may not be reliable. The statistical analysis shows that our bacterial counts for TC and *E. coli* are statistically the same as the bacterial counts obtained by the county. This result suggests that the multiple tube fermentation method yields the same result as the IDEXX Quanti-Tray™ enumeration method, and also validates our laboratory analysis result. The equivalence of our sample analysis with the county's analysis enables us to proceed with further extrapolation from the results.

Our results show that the mean TC count for samples with sediment is 4.27 times higher than the mean TC count for samples without sediments (See Appendix V). The mean result of *E. coli* count for samples with sediment was 1.33 times higher than the mean result of samples without sediments. However, this difference for *E. coli* is not statistically significant, which is contrary to results from past research. Experiments by Van Donsel and Geldreich (1971), Hood and Ness (1982), LaLiberte and Grimes (1982), Ghoul *et al.* (1990), Marino and Gannon (1991), and Davies *et al.* (1995) all have shown that fecal coliform is significantly higher in sediments.

Bacterial loading from re-suspended sediments may be a consequence of bacterial community desorption from sediment particles. At the microscale, there are three mechanisms proposed to explain the interaction between bacteria and particulate matter: 1) bacteria adsorb to the surface of larger particulates, 2) bacterial aggregates coflocculate with soil particles of similar size, and 3) colloidal particles adsorb to bacterial surface (Marshall 1980). The stability of the particle-bacteria aggregates is dependent upon the valency of the dominant cations in the system. Increased cation concentration leads to increased sorption of bacteria to particles (van Loosdrecht *et al.* 1989).

The microorganisms can be reversibly sorbed or permanently adhered to the particulate matter (Marshall 1980). When bacteria are reversibly sorbed, they are not anchored to the particle surface. Therefore, the application of shear force may cause the bacteria to desorb. Permanent adhesion occurs when bacteria are anchored to particulates and remain attached even when shear force is applied. Our sediment experiment demonstrates that bacteria may be reversibly sorbed to the sediment. Thus, when force is applied (e.g. stirring of sediment artificially or by increased flow due to rain events), the bacteria detach from the sediment surface and are released into the water column. Also, bacteria may be more numerous in interstitial water. An experiment by Roper and Marshall (1974) shows that the number of bacteria sorbed on sediment increases as a function of increased salinity. Therefore, when bacteria reach the ocean from the freshwater streams, sorption and/or flocculation increases. There is evidence that when bacteria are sorbed on to surfaces, the efficiency of bacterial predators such as planktonic protists (Iriberri *et al.* 1994) at killing bacteria is reduced (Roper and Marshall 1974). The implication of this result is that there may be an accumulation of bacteria in marine sediment. When bacteria are transported from creeks into the ocean, marine sediment may serve as a reservoir for potentially pathogenic bacteria. Based on our health risk analysis, (see section 4) the health risk to beach users increases during storm events when bacteria in the creeks increase as a result of loading from sediments and surface runoff.

In a storm event, bacteria sorbed on the sediment may desorb and partition into the water column. In order to quantify the amount of bacteria loading from the sediment, the sorption constant of bacteria to sediment (K_s), and concentration of bacteria (C_w) in the water column is needed. Using the relationship $K_s = C_s/C_w$, the concentration of bacteria sorbed on the sediment (C_s) can be calculated, where the unit is MPN/mass of sediment. The Roper and Marshall study (1974) demonstrates reversible sorption of natural and artificially added *E. coli* in saline montmorillonite sediments using a stepwise washing technique. Following each washing and centrifugation of the sediment, the number of microorganisms in the supernatant is determined. Their result shows that after washing, the number of bacteria in the supernatant is higher than the artificially added number. The higher number of bacteria may be a result of indigenous microorganisms desorbed from the sediment after each washing. If we assume that the amount of bacteria loading due to storm events is equivalent to the amount of bacteria sorbed on the sediment, and if we know the amount of sediment

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increase due to storm events, we can approximate the total mass of bacteria loading into the ocean from the sediment. The amount of sediment transported during storm events is dependent on the geological characteristics of the watershed, the amount of precipitation, and the volume of flow within the stream channel (Dunne 1999). This will be discussed further in the hydrology section.

Mills *et al.* (1994) found that the K_s for bacteria sorbing on to quartz sand ranges from 0.55 to 6.11 mL/g. Their research also indicates that the threshold for bacteria sorption onto iron-oxyhydroxide-coated sand is 6.93×10^8 cells/g of coated sand. The amount of bacteria sorbed on to sediment may vary according to sediment particle size, sediment particle type, organic content of sediment, salinity, pH, etc. Hoppe (1984) reports that in beach sand at 1 cm of depth, the total number of bacteria in the column is 116×10^6 cells/mL, and the number of sorbed bacteria is 106×10^6 cells/ cm^3. The percentage of the sorbed bacteria is 90%, indicating that most of the bacteria in marine environments are sorbed. Numerous parameters, including the K_s, will contribute to variability in the calculated amount of indicator organism loading. However, an approximate range of the indicator bacteria MPN will still aid in indicating the possible range of increases in health risk. The amount of indicator bacteria loading from the sediment into the water column during a storm should be taken in account when formulating and implementing monitoring regulations in order to protect best public health.

In our sediment experiment, we did not compare enterococcus levels in samples with sediments and samples without sediments. We also did not compare the indicator bacterial levels in ocean samples with and without sediments. It would be interesting to compare the results of enterococcus counts in ocean samples with our sampling results. Enterococcus is known to survive longer in marine water (Anderson *et al.* 1997), and a higher population may also survive in the sediment than TC and FC. A correlation between the last sampling point in the creek and the point where marine water mixes with the freshwater is expected. However, after performing a statistical analysis on Project Clean Water data provided by Santa Barbara County Water Resources (Oct 98-Nov. 98), a clear correlation between the indicator bacterial counts at the last point in the creek and the bacterial counts in the ocean water could not be found. Appendix VI shows that there is no statistically significant correlation between either total coliform count or *E. coli* count in the sample from creek outfall and from the ocean; however, the outfall levels are higher for all. Temperature, tidal, salinity, and sampling location variability all contribute to bacterial counts in the ocean; thus, a simple linear correlation does not exist. Even though creeks are major sources of bacteria, other factors such as waders, birds and septic tanks may lead to high bacterial levels in the beach water.

In the past, the county of Santa Barbara did not sample for indicator bacteria during heavy rainfall due to the inability of Quanti-Tray™ to interpret muddy samples. It is important to determine the bacterial levels during rain events because of the high levels associated with rainfall as shown through our statistical analysis and sediment

experiment. Therefore, we suggest that the county laboratory utilize a bacteria quantification method for turbid samples for the benefit of implementing appropriate beach closure to protect public health.

2.2.3   Fertilizer statistical analysis

Methods

Since many environments are nitrogen-limited for bacteria, we hypothesized that nitrogen addition would have the effect of increasing the number of bacteria in the system. Fertilizer containing nitrogen was applied at Baron Ranch, located in the Arroyo Quemado watershed during the summer of 1998. In order to test whether the fertilizer had an effect on bacteria counts, we performed a paired two-sample Student's t-Test for means at α=0.05 (Ott 1993). We performed this test on total coliform counts for a site on Arroyo Quemado Creek below all of the fertilizer additions. Bacterial data from three months prior to fertilization were used for the unfertilized sample, and data from three months after the first fertilization date were used for the fertilized sample (Santa Barbara County Solid Waste &Utility Division. Mar. 1997-Sept. 1998). The null hypothesis was that there was no significant difference between the means of the two samples. The alternative hypothesis was that the unfertilized sample has a lower mean than the fertilized sample. We compared the means of 10 observations for each sample, after checking for normality and removing one outlier.

Fertilization analysis results

Table 2.1

Test Period	1998
Mean TC count for unfertilized sample (MPN/100mL)	2914.6
Mean TC count for fertilized sample (MPN/100mL)	7740.8
Variance unfertilized sample	4310169
Variance fertilized sample	23195742
Degrees of freedom	9
T critical for α=.05	1.83
T statistic	-2.73
P(t)	.01
Reject null hypothesis?	Yes

We can reject the null hypothesis for 1998, as seen in the table. In order for the unfertilized sample to have a significantly lower mean than the fertilized sample, the t statistic must be less than the absolute value of t critical, and the p value must be less than .05. Both of these factors are true for this data set. Therefore, there were more coliform bacteria present after fertilization than before fertilization.

Discussion of fertilization analysis

Our analysis supported our hypothesis that higher levels of bacteria would occur after fertilizer addition. However, since the unfertilized sample occurred during the spring months and the fertilized sample occurred in the summer months, this difference may be seasonal rather than due to the fertilizer addition. This difference would contradict our earlier hypothesis that higher levels of bacteria occur during precipitation events. In order to corroborate this finding it would be necessary to repeat this test with both bacteria data and nutrient data. Since nutrient data were not available for the 1998 time period, this was not possible.

2.2.4   High levels of bacteria during the dry season

Loading from the sediments into the water column is only one of the sources of bacteria in ocean water. During the dry season of 1997 (sampling dates 3/24/97-8/24/97), Arroyo Burro was out of compliance for total coliform 2 out of 24 weeks (SB County EH&S). During this same season, Arroyo Quemada was out of compliance 19 weeks (Ibid). During the summer of 1998 (5/26/98-8/31/98), Arroyo Burro was out of compliance for total coliform for 15 out of 15 weeks. Arroyo Quemada was sampled only 9 weeks during this period, and it was out of compliance for total coliform 8 of those weeks.

Several factors may be contributing to the high levels of bacteria found at these sites. Temperature plays a major role in bacterial survival and propagation. Ramteke (1995) and Markosova and Jezek (1994) fit regression curves and found a significant positive correlation between coliform counts and water temperature. Baudisova (1997) found that the total coliform counts in the summer "increased by <2 orders of magnitude." The ideal temperature for *E. coli* is 39°C (Madigan *et al.* 1997) and the ideal temperature for many enterococci species is 44°C.

Another possible cause of high bacterial concentrations in the summer months may be the increased numbers of swimmers. Papadakis *et al.* (1997) found that that yeasts of human origin correlated with the numbers of bathers at the site. It would be interesting to investigate whether the number of indicator organisms found had any correlation with the number of swimmers at a beach. It would be possible to conduct this study in Santa Barbara.

In the summer of 1998, when no rainfall occurred, the bacterial levels at Arroyo Quemado and Arroyo Burro were high, and one possible reason is due to gull droppings into the beach water. Close to 5,000 gulls (Winant 1998) have been observed at Arroyo Quemada Beach, which leads to a significant amount of droppings into the water. Several studies have demonstrated that droppings of avian species lead to high bacterial levels in the water (Ortiz and Smith 1994; Oshiro and Fujioka 1995; Wyer *et al.* 1995b; Anderson *et al.* 1997). The range of enterococci reported for per

gram of gull droppings is 5.13 x 10^2 – 3.48 x 10^4 cfu (Anderson *et al.* 1997). The Anderson study (1997) found that bird droppings from ducks and gulls exhibit higher enterococci count than mammals, such as cattle and sheep. Oshiro and Fujioka (1995) found that "pigeon feces, per gram, contained 1.6 x 10^8 fecal coliform, 1.7 x 10^8 *E. coli*, and 4.0 x 10^5 enterococci." They also analyzed shoreline, water, sand, land runoff, and mongoose and pigeon droppings for fecal coliform, *E. coli*, and enterococci. They found that the major source of beach sand contamination was pigeon feces. Although this study was conducted in a watershed with few sources, a modified study of this type could be undertaken in Santa Barbara County at both Arroyo Quemada and Arroyo Burro.

Tidal variation may play a role in the bacterial levels in ocean water. The study by Koh *et al.* (1994) found significant differences between samples of the pathogenic genus *Vibrio* taken at different tidal levels of concentrations. The researchers found that the highest concentration of *Vibrio* occurred at low tide. The speculated reason was that the "ebbing tide stirs up the bottom sediments, so that the high concentrations at low tide could be a reflection of the increased numbers of *vibrios* that were re-suspended from the sediments" (Koh *et al.* 1994). It is possible that levels of total coliform, *E. coli*, and enterococcus also fluctuate with regard to the tide level and that variations in temperature and salinity may be factors in ocean bacterial levels.

2.3   Summary of microbial ecology results

Research indicates that possible sources of bacterial pollution, among others, include septic tanks (Wyer *et al.* 1997), guano (Oshiro and Fujioka 1995), cattle grazing (Howell *et al.* 1995), and bathers (Papadakis *et al.* 1997). In order to determine the pathogenicity of the ocean water at these sites, it is important to know the sources of the bacteria. For example, if the major source is found to be birds, then the water would be significantly less pathogenic than if the source was a leaky sewage line.

Use of other indicators other then TC, FC, and enterococci may be necessary in these watersheds to determine whether human feces are the primary source of pollution. As mentioned earlier, sorbitol-fermenting bifidobacteria could be used since this indicator is specific to human waste. Molecular methods that allow for specific targeting of human and other feces might also be useful.

Research by Howell *et al.* (1995), Wyer *et al.* (1995b), and Jagals *et al.* (1995) concluded that an increase in rain results in an increase in indicator bacteria levels. Our sampling results clearly suggest a positive correlation between bacteria counts and precipitation during the rainy season. Our statistical analysis shows that precipitation has a correlation coefficient of .75 to bacterial counts in the ocean. Rain events may lead to increased bacterial levels in creeks and the ocean via two mechanisms: 1) fecal matter on dry land is washed into the creeks and storm drains by the rain water and 2) sediments are re-suspended and bacteria are released into the water column. Our experiment attempted to simulate rain events that would re-

suspend the sediment in order to elucidate the reason for increased bacterial levels during rain events. Even though the scale of sediment re-suspension in our experiment is miniscule compared to the amount of sediment re-suspended during an actual storm event, and the experiment was performed on creek sediment, the results are expected to be similar in beach water, according to past research (Wyer *et al.* 1995b; Roll and Fujioka 1997). Sediment re-suspension caused by events such as rain, waders in the recreation water, and motor boats, is a major source of indicator bacteria in the water column. Therefore, public health risk increases when sediments are stirred and bacteria are released. Future government regulations need to incorporate monitoring of the loading of bacteria from sediments in order to fully protect public welfare.

Bacteria persist everywhere in the natural environment. Therefore, there is a baseline bacterial level in the water, and natural fluctuations occur for bacterial populations due to factors such as temperature, sunlight, nutrients, oxygen availability, pH, presence of predators, etc. Contrary to general perception, not all bacteria are pathogenic and harmful to human health; they are an integral component of ecosystems. Therefore, remediation events should take in account of the ecological importance of indigenous bacteria before implementing complete extermination of bacterial levels in the creek and ocean water. Remediation techniques will be addressed in the best management practices (BMPs) section of this report.

3.0   Hydrology

3.1   Background

In order to understand bacterial pollution at Arroyo Burro and Arroyo Quemada beaches, we must examine the hydrology of their watersheds. In addition to point sources such as sewage spills, coastal bacterial pollution is the result of stormwater runoff carrying pollutants generated by nonpoint sources within the watershed. The amount of pollutants contained in storm water runoff will vary widely based on the physical and chemical characteristics of the various land uses within the watershed. For example, increased areas of imperviousness associated with urban development will increase the amount of storm water runoff (Ventura and Kim 1993). In this section of the report, we demonstrate the effect of imperviousness on the amount of stormwater runoff, and draw some conclusions about the resulting increase in sediment and bacterial loading to coastal waters.

In order to determine the impact of land surface development on watershed and coastal water health, we employed the Soil Conservation Service's Rainfall-Runoff Depth Relation Model. This model has enabled the group to predict approximate expected runoff volumes from the Arroyo Burro and Arroyo Quemado watersheds, given a certain precipitation event. The Arroyo Burro watershed contains a variety of land uses and is representative of an urban watershed. The Arroyo Quemado watershed is less developed and represents a more rural watershed. We created a GIS

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platform for the Arroyo Burro watershed in order to obtain the input parameters for the SCS model. We used analog maps for the Arroyo Quemado watershed to obtain these parameters. The following is a description of both the Arroyo Burro and Arroyo Quemada study sites.

3.1.1 Regional Climate

Santa Barbara has a Mediterranean climate with mild temperatures and seasonal rainfall. The average annual temperature is 15.6°C (NOAA 1994). Summer months have very little rainfall, with fog and low cloudiness occurring in the coastal zones at night. Precipitation is concentrated during the winter months from November to April (USDA 1981). Because of the Mediterranean climate and variable topography, the region experiences a significant orographic precipitation effect, with the majority of precipitation falling in the mountains, and less rain falling on the coastal plain. (NOAA 1994) The mean annual rainfall for the City of Santa Barbara is approximately 17 inches (SB County Flood Control 1998). The amount of rainfall in 1997-98 year was much higher (46 inches) than in an average year because of El Niño (Ibid).

3.1.2 Study Sites

Arroyo Quemada Beach

There are two watersheds that discharge at Arroyo Quemada beach: Arroyo Quemado creek and Pila creek. The Canada de la Pila (Pila Creek) Watershed, is approximately 415 acres in size (EMCON 1994). Surface water flow in Pila is ephemeral during short periods during and following heavy precipitation events (EMCON 1994). In 1967, the County of Santa Barbara constructed the Tajiguas Landfill, which occupies 130-acres in the southern portion of the Pila Creek Watershed, across the Canada de la Pila stream valley. Water is diverted around the west-side of the landfill through a 48" diameter culvert and is discharged below the landfill (Ibid). The other land uses in this watershed are open space and agriculture.

We focused on the Arroyo Quemado watershed in modeling runoff. Arroyo Quemado creek is a perennial stream that drains a watershed of approximately 1250 acres. (EMCON 1994) The Arroyo Quemado Watershed is located immediately to the east of the Pila Watershed. There is a small lagoon at the end of the creek and a sand berm is usually present during periods of low flow, separating the lagoon from the ocean. Primary land uses in the watershed include Baron Ranch, a County owned and operated avocado orchard, and National Forest Lands, which also support limited cattle grazing. There is a community of 10 developed parcels located southeast of the landfill, along Arroyo Quemado Lane. These parcels are located near or on the bluff overlooking the ocean and Arroyo Quemada beach. All parcels in the community are on septic systems. There are two parcels immediately to the west and one to the east

of the Arroyo Quemado creek, on either side of the lagoon. Table 3.1 describes the land use within the Arroyo Quemado watershed.

Table 3.1 Arroyo Quemado Watershed Land Use Categories

Land Use	% WS	Area (Acres)
Residential	3.0	37.5
Agricultural	15.0	187.5
Natural Vegetation	80.0	1,000.0
Roads	2.0	25.0

Arroyo Burro Beach

The Arroyo Burro Watershed drains approximately 6,289 acres. The creek and its tributaries are perennial. The main tributaries are San Roque Creek and Las Positas Creek, which join Arroyo Burro just north of U.S. Highway 101. Arroyo Burro creek flows southward and forms a lagoon at Arroyo Burro Beach. A sand berm forms during periods of low flow that prevents the creek from reaching the ocean.

National Forest Lands, agricultural lands, and some low-density residential areas dominate the upper reaches of the watershed above Foothill Boulevard. Between Foothill Boulevard and U.S. Highway 101, primary land uses include commercial and medium to high density urban residential. A golf course and the Earl Warren Showgrounds lie just west of Las Positas Road. There is a similar scheme below the freeway and to the beach, with the exception of the low-density residential homes in Hope Ranch, and Las Positas Park. Table 3.2 provides the land use distribution for the Arroyo Burro watershed.

Table 3.2 Arroyo Burro Land Use Categories

Land Use	% of the Watershed	Area (Acres)
Commercial	5.17	326.5
County	11.72	740.1
Golf Course	1.98	124.9
Natural Vegetation	46.41	2,930.5
Laurel Canyon Dam	0.99	62.2
Residential <.25 acre	14.96	944.5
Residential = .5 acre	3.89	245.5
Residential > 1 acre	8.20	517.8
Multiple Residence	0.27	17.2
Agricultural	1.70	107.6
Roads	4.71	297.6
Totals	100.00	6,314.4

3.2 Construction of a Geographical Information System (GIS) Platform for the Arroyo Burro Watershed

In order to aid investigation of coastal bacterial contamination at Arroyo Burro Beach, we developed a detailed representation of the study area and associated watershed. Initially, we used analog maps of the watershed and field trips to familiarize ourselves with the study site. With time however, it became possible to develop a comprehensive Geographical Information System (GIS) platform. The GIS platform provides the ability to:

- Calculate the area of identified land uses within the watershed. This then enabled us to run a rainfall-runoff model to approximate the expected storm runoff at Arroyo Burro Beach, given a certain precipitation event.

- Aid others in a spatial analysis of non-point source pollution to determine if a correlation exists between high indicator organism concentrations and surrounding land use.

In order to develop our GIS platform, we used the ArcView (Version 3.1) and ArcInfo software packages developed by the Environmental Science Research Institute (Redlands, CA, USA).

Once we collected the data, we incorporated it into a consistent format, or projection. In some instances, this consisted of changing the projection of the data from

projection "x" to the "Albers Equal-Area Projection," used with this platform. We did this using ArcInfo, which is software that stores features and their attributes in its own format. In order to incorporate data only relevant to Arroyo Burro, we "cut" or fit data layers using ArcInfo to trim the coverages to the watershed. We used ArcView to create the stream network within the watershed. Using 30-meter resolution digital elevation maps (DEM) obtained from USGS and the commands called flowaccumulation and flowdirection, we derived grid cells with related flow accumulation values as streams. Once we obtained data, incorporated it into a uniform projection, and confined the data to the Arroyo Burro watershed boundaries, we then incorporated it into one GIS platform. Appendices VII and VIII contain maps illustrating the GIS for the Arroyo Burro study site.

Data Layers

Table 3.3 summarizes the sources of the identified data layers for the Arroyo Burro watershed needed to construct the GIS platform.

Table 3.3 GIS Data Sources

Data Layer	Source
Road network	Penfield & Smith 1992
Stream network	USGS DEM's 1998
Land use	Penfield & Smith 1992
Land use	USGS 1988a and 1988b
Soils	USDA 1981
Watershed boundary	USGS DEM's 1998
Sampling points	Santa Barbara County Water Resources 1999
Sewer lines	Santa Barbara County Water Resources 1999

Road Network

The Arroyo Burro watershed contains approximately 110 kilometers of roadway. The approximate area of this road network is 1,190,510 square meters. After changing its projection to Albers Equal-Area Projection, we incorporated the road network data in its raw form. As can been seen in the platform, the majority of the roads are found within the bottom half of the watershed. While the total area of the watershed covered by roadway is relatively small, roadways contribute a disproportionately high percentage of runoff during storm events with dry and moist antecedent soil conditions due to the high curve number associated with them (See SCS Model).

Stream Network

Arroyo Burro beach is bisected by Arroyo Burro Creek. Arroyo Burro Creek is a third-order creek. Primary tributaries are Las Positas Creek and San Roque Creek. The Arroyo Burro stream network was developed using USGS 30-meter digital

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elevation maps and the GIS program ArcInfo. No attribute data was available for computing the length of the stream network. However, for modeling purposes, this was acceptable. The percentage of land covered by the stream network was deemed insignificant when compared to the watershed as a whole.

Land Use

The land use data as obtained from the City of Santa Barbara only covered approximately 65% of the Arroyo Burro watershed. However, this data layer covered the major urban areas within the watershed. To fill in the remainder of the data for the land use layer, we used USGS orthographic quads in conjunction with site inspections. When the City data was first incorporated, it contained a level of detail too fine for our purposes. It became prudent to combine some categories of data. We used the land use types as described in the Soil Conservation Service's Curve Number table to compartmentalize the City land use descriptions into general land use categories that would enable quick analysis for modeling purposes. The land use categories found within the Arroyo Burro watershed are presented in Table 3.2.

It can be seen that the primary land use designation within the Arroyo Burro watershed is natural vegetation. These areas are associated primarily with the Los Padres National Forest. County lands consist primarily of open spaces (Las Positas Park) and limited (less than 10%) residential plots of size <0.25 acres. While the majority of this area is associated with parks and open spaces, small residential units occupy a small percentage of this area. It should be noted that due to overlapping land use designations in some areas, the total watershed acreage presented using the GIS platform (6,314.4 acres) is slightly greater than the true value (6,289).

Soil Type

The soil layer was generated using data compiled by the USDA Soil Conservation Service's survey of Santa Barbara County in 1981. The soil types are broken down into categories based on the infiltration capacity of the soil.

Watershed Delineation

The watershed was delineated using USGS 30-meter digital elevation maps and ArcInfo.

Sampling Points/Sewer Lines

We obtained sampling points and sewer line locations from the County of Santa Barbara. The sampling points are from Project Clean Water and the South Coast Watershed Characterization Study. Sewer line crossings are areas where sewer lines bisect Arroyo Burro Creek or one of its tributaries.

3.3   Stormwater Runoff Model

3.3.1   The Soil Conservation Service (SCS) Rainfall-Runoff Depth Relation Model

The Soil Conservation Service developed The Rainfall-Runoff Depth Relation Model in 1972. We selected this technique because it is reliable, has been used for many years in the United States, is computationally efficient, and the required inputs are generally available. The model predicts the volume of storm runoff for a small watershed based on the depth of rainfall, the land cover/use type, the soil type, and the antecedent soil moisture condition. The following description of the SCS model was adapted from *Hydrologic Analysis and Design* (McCuen 1989).

The depth of rainfall (P) can be divided into three components: direct runoff (q), initial abstraction (I_a), and actual retention (F). Direct runoff is the storm runoff resulting from excess rainfall falling on an area. Initial abstraction is the amount of rainfall that is initially absorbed for a certain area, given a specified rainfall event and for certain surface and subsurface conditions. Actual retention is the amount of rainfall minus the initial abstraction and the direct runoff, for a certain area. Empirical evidence has suggested that Ia = 0.2S, where S is defined as the potential maximum retention for a land cover surface. S is a function of land use, interception, infiltration, depression storage, and antecedent moisture. After some algebraic manipulation, this results in the SCS
Rainfall-Runoff Relationship Model:

$$q = (P - .2(S)^2) / (P - .8(S)) \qquad \text{(Equation 3.1)}$$

Where S = (1000 / CN) – 10.
And CN = Curve Number for the area.

It is important to note that P ≥ .2(S), in order to generate q.
Therefore, q = 0 when P < .2(S).

In order to predict runoff (q) from a known storm event, the one unknown in the equation, S, must be estimated. In order to do this, the SCS runoff Curve Number was developed. The curve number is a dimensionless numerical representation (0-100) of the infiltration capacity of a surface. Surfaces that have a high infiltration capacity have a low curve number. These surfaces will be expected to produce less runoff than that of a surface with a high curve number and a low infiltration capacity, for example, a parking lot. The curve number, or CN, is an index that is a function of three factors: (a) hydrologic soil group, (b) land cover/use type, and (c) antecedent moisture conditions of the soil. Depending on the land use, treatment, and hydrologic condition, separate curve number values are given for each soil group. To see a complete list, please refer to Appendix IX.

The SCS developed a soil classification system that consists of the four groups shown in table 3.3.

Table 3.3 Hydrologic Soil Groups

Group	Type	Infiltration (inches/hr)
A	Deep sand, deep loess, aggregated silts	0.30-0.45
B	Shallow loess, sandy loam	0.15-0.30
C	Clay loams, shallow sandy loam, light clays	0.05-0.15
D	Swelling soils, heavy clays, saline soils	0.00-0.05

Source: McCuen 1989

The Soil Conservation Service (USDA) determined these infiltration rates through surveys conducted in 1977. The Arroyo Burro Watershed consists primarily of soils of type A, B, and C. These soils are associated with high, medium, and low infiltration rates, respectively. The Arroyo Quemado watershed consists primarily of soil types B, C, and D.

The volume of runoff from a soil is strongly dependent upon the antecedent soil moisture content. For example, if a one inch storm were to fall early in the rain season when the soil is mostly dry, runoff would be expected to be much less than if the same one inch event were to fall following a month of wet weather. Realizing this, the SCS developed three antecedent soil moisture conditions:

Condition I: Soils are dry but not to their wilting point; satisfactory cultivation has taken place.
Condition II: Average conditions.
Condition III: Heavy rainfall, or light rainfall and low temperatures has occurred within the last 5 days; saturated soil.

Table 3.4 Total 5-day Antecedent Rainfall Thresholds for ASMC

ASMC	Dormant Season (inches)	Growing Season (inches)
I	<0.5	<1.4
II	0.5-1.1	1.4-2.1
III	>1.1	>2.1

Source: McCuen 1989

For both the Arroyo Burro and Arroyo Quemado watersheds, the dormant season can be defined as the months from October through March, with the growing season taking place from April to September. Because there is little agriculture within the Arroyo Burro watershed, there is little difference between the two conditions.

3.3.2   Methods

Arroyo Burro

Once we incorporated all of the relevant data into the GIS platform, we were able to begin interpreting the physical layout and description of the Arroyo Burro watershed. Using attribute data tables associated with data layers in the GIS platform, we were able to discern the approximate areal extent for identified land uses within the watershed. Following this, we overlaid the hydrologic soils grouping layer with the land use layer to determine the type of soil underlying each identified land use within the watershed. Then, using the identified land use and associated hydrologic soil group, we assigned curve numbers to each identified land use group. Curve numbers were developed assuming an average, or Condition II antecedent soil moisture content level. In areas where the same land use overlaid different soil groups, we employed an average curve number value that reflected the changes in hydrologic soil types throughout the watershed.

Having determined curve numbers for each land use for average antecedent soil moisture conditions, we then assigned curve number values for the identified parcels for both a dry antecedent soil moisture conditions (Condition I), and a wet antecedent soil moisture conditions (Condition III).

Being situated in a Mediterranean climate regime with variable topography, the Arroyo Burro and Arroyo Quemado watersheds do at times experience a significant orographic effect, with the majority of precipitation falling in the mountains, and less rain falling on the coastal plain.   It can be expected that for any given winter season storm, more rain will fall at elevation (the upper reaches of the watershed) than at the coast. This additional rainfall at elevation could translate into more runoff from the higher elevations of the watershed, depending on land use, soil type, and antecedent soil moisture conditions. In order to determine the orographic effect of precipitation on the watershed, the amount of rainfall (P), has been altered to reflect expected precipitation at various elevations, using developed historical isohyetals and observed rainfall readings obtained from the County of Santa Barbara (SB Flood Control 1975). The isohyetals depict that annual average precipitation for lower elevations in the watershed is expected to range between 16-20 inches, while the upper reaches of the watershed is expected to receive annual average rainfall of 26-30 inches. Therefore, precipitation in our model is expected to be approximately 65% greater at elevation than at sea level. A one-inch precipitation value is assigned to the lower elevations (below Foothill Drive), while a 1.65-inch precipitation event has been applied to the higher elevations of the watershed (above Foothill Drive to the drainage divide.)

Arroyo Quemado

The methods used to model runoff for Arroyo Quemado were the same as those used for Arroyo Burro with the exception of data sources. While we used a GIS platform to

estimate land use cover and soil type for the Arroyo Burro watershed, we used analog maps for the Arroyo Quemado watershed. We used four USGS 7.5 minute topographic quads (USGS 1988b) and a land use overlay from the UCSB Map and Imagery laboratory to approximate land use areas for the watershed. The soil types associated with each land use were estimated using these two maps and overlain with soil maps from the US Department of Agriculture Soil Survey (USDA 1981).

3.3.3   Results

Arroyo Burro

For our model purposes (a one-inch event), when S is greater than or equal to 5, no runoff will occur. In addition, when S is greater than or equal to 7.75 for P = 1.65, q = 0. Inspection of the model results reveals that in a few instances (predominately in dry antecedent soil conditions), S is greater than 5 or 7.75, resulting in a zero value for q for that land use type.

Using Microsoft Excel spreadsheets, we ran the SCS Rainfall-Runoff Model for the Arroyo Burro watershed. As described above, q is the depth of stormwater runoff for a given precipitation event (P), expected from each designated land use. This depth, q, in inches, was then converted to a depth in meters, and then multiplied by the surface area for each respective land use scheme. This returned an approximate expected runoff volume, Q, in cubic meters, for each land use scheme within the watershed. By adding up the contribution from each land use runoff volume, a total approximate Q_{tot} ($Q_{tot}=\Sigma Q$) for the Arroyo Burro Watershed was determined. See Table 3.5 below for a summary of the model predictions.

Table 3.5 SCS Rainfall-Runoff Model Predictions: Arroyo Burro Watershed for P=1"

ASMC	Runoff (Q_{tot}) in m^3
I (Dry)	23,700
II (Moist)	58,400
III (Dry)	220,000

Arroyo Quemado

The Arroyo Quemado watershed is similar in topography to the Arroyo Burro Watershed. However, the Arroyo Quemado watershed is underlain by less permeable soils in the upper reaches, and by soils of medium permeability near the shoreline and along the creek. Additionally, the primary land cover within the watershed is natural vegetation.   Given the same magnitude precipitation event, stormwater runoff values have been modeled for the 1,250-acre watershed. Given the steep topography of the watershed, the same assumptions used for precipitation distribution at Arroyo Burro

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were used at Arroyo Quemado. Using USGS topographic quads and the Soil Service soil survey findings, stormwater runoff results were generated for the Arroyo Quemado watershed using Microsoft Excel spreadsheets within the SCS Rainfall-Runoff Model.

Table 3.6 SCS Rainfall-Runoff Model Predictions: Arroyo Quemado Watershed for P=1"

ASMC	Runoff (Q_{tot}) in m^3
I (Dry)	4,120
II (Moist)	38,100
III (Wet)	114,000

For a summary of the total expected runoff as a function of antecedent moisture conditions in each watershed, please consult Appendix X.

3.4   Discussion and Conclusions

The SCS Rainfall-Runoff Model produced approximate storm runoff volumes for existing land use conditions and various antecedent soil moisture conditions within the Arroyo Burro and the Arroyo Quemado watersheds. While we only ran this model for a one-inch storm in the lower elevations of the watersheds, it is possible to change the precipitation values to reflect a different sized rainfall event.

When a storm of a given intensity occurs, with given antecedent soil moisture content and soil and land use conditions, a certain level of stormwater runoff can be expected. As approximated by the model, certain meteorological and physical conditions can have a drastic effect on stormwater runoff values. A storm arriving in the beginning of the rainy season should result in less stormwater runoff than a storm of the same size and intensity occurring later in the water year. This is reflected in the model's prediction for more runoff in wet conditions. Depending on the source of indicator bacteria and the amount of predicted stormwater runoff, indicator bacterial concentrations at Arroyo Burro and Arroyo Quemada beach will be affected. While high stormwater runoff allows for the potential for more indicator bacteria to reach coastal waters, it also lowers indicator bacteria concentrations through dilution. Conversely, indicator bacteria loading levels can be low for a small runoff event, but the concentration can be high.

As detailed in the SCS rainfall-runoff model discussion, the value of the curve number has a significant effect on runoff. Therefore, it can be expected based on simple analysis of the formula, that as the curve number decreases, the surface retention decreases, leading to decreased storm runoff, given a certain precipitation event. The Arroyo Quemado watershed has lower curve numbers than Arroyo Burro because it has less urban development. As a result, the model predicts less runoff for an identical

precipitation event. Table 3.7 summarizes the differences in stormwater runoff from the two watersheds given an identical precipitation event. Appendix X illustrates these differences graphically.

Table 3.7 Stormwater Runoff (Q_{tot}) from Arroyo Burro and Arroyo Quemado with P=1":

ASMC	Arroyo Quemado	Arroyo Burro
I (Dry)	4,120	23,700
II (Moist)	38,100	59,400
III (Wet)	114,000	220,000

The results of the SCS model suggest a noticeable effect of urbanization on watershed stormwater runoff, given a certain precipitation event. For an identical precipitation event during dry ASMCs, the effect of urbanization is shown in the high volume of stormwater runoff from the Arroyo Burro watershed. While the Arroyo Burro watershed is approximately 5 times larger (6,289 vs. 1,250 acres), stormwater runoff is approximately 6 times larger (23,700 vs. 4,120 cubic meters). In contrast, the effect of the lower permeable soils underlying the natural vegetation within the Arroyo Quemado watershed is reflected in the large predicted stormwater runoff value for wet soil conditions. While the Arroyo Quemado watershed is much smaller, the lower permeable soils underlying this area produce a predicted stormwater volume that is greater than half that of Arroyo Burro for wet soil conditions (114,000 vs. 220,000 cubic meters).

Unfortunately, an active stream gauging station does not exist anywhere within the Arroyo Burro or Arroyo Quemado watersheds. However, the USGS did have an active stream gauging station in the Arroyo Burro watershed located near the confluence of Arroyo Burro and San Roque Creeks for the period from 1970 to 1993. While this investigation did not attempt to calibrate the model watershed runoff with historical values, there exists the potential to do so. Historical stream flow volumes from the USGS station, in conjunction with historical rainfall data obtainable from the County of Santa Barbara would enable higher resolution rainfall-runoff modeling of the upper reaches of the Arroyo Burro watershed.

Sediment Transport

We have shown that as the amount of precipitation within a watershed increases storm runoff volume also increases. As storm runoff volume increases, sediment transport can also increase (Dunne 1999). Sediment particles adsorb many contaminants that are transported, deposited and stored as part of the sedimentary component of the river system (Meade 1984). The microbiology section has shown that indicator bacteria, like other contaminants, can sorb onto sediment particles. It is possible that these indicator bacteria could desorb during a rain event and contribute to indicator bacterial

loading to the water column. We therefore conclude that increased sediment loading due to increased runoff is a significant mechanism for indicator bacterial transport.

Runoff modeling of the Arroyo Burro and Arroyo Quemado watersheds has predicted that discharge at their respective beaches will be small for a one-inch rainfall event with dry antecedent soil moisture conditions. The likelihood of large amounts of sediment being transported in this event is small, with all other conditions remaining the same. However, if this is an identified "first-flush" (first significant rain event for the rainfall season), the runoff can contain significant amounts of sediment. As precipitation and the antecedent soil moisture content increase, there exists the potential for creeks to deliver increased sediment loads to the ocean. For extreme precipitation events, it is not uncommon to see many years' worth of sediment being transported in just a few days (Dunne 1999). Also, over a period of many years, a large proportion of the long-term sediment load may be transported in response to a few large, but infrequent, storms (Meade 1984). Based on our conclusion that indicator bacterial loading increases with sediment loading, large amounts of indicator bacteria could be transported as well.

Total Loading

It is possible to determine the approximate total loading of coliform bacteria at Arroyo Burro Beach given a certain precipitation event. By multiplying observed indicator bacteria concentrations (sediment and suspended concentrations) from watershed locations with the expected runoff volume for a given precipitation event, an approximate value for total indicator bacteria loading can be predicted for a particular stream location and its associated land uses. Done at many locations within the entire watershed, these calculations can then be summed in order to predict total loading. Such a prediction could aid future investigators and policy makers in their decision-making processes for dealing with the challenges associated with coastal bacterial pollution.

4.0   Analysis of Potential Health Risks

Assessment of environmental health risks requires an understanding of how the processes that govern pollutant loading affect the probability of human exposure to chemicals or pathogens. Ultimately, we are interested in examining what possible adverse health effects result from coming in contact with pollutants in our environment. Generally, when performing an environmental health risk analysis, there are four key steps in determining the likelihood of adverse health effects to a population:

(1) *Risk identification*. This step defines the risk. For example, swimming in polluted water can result in contracting certain illnesses.

(2) *Dose/response assessment.* This step realizes that exposure to specific quantities or concentrations of a contaminant has specific health effects. Epidemiological data provide insight into this dose/response relationship.

(3) *Exposure assessment.* This requires scientific knowledge in order to determine the transfer coefficients of the pollutants through different media, and the likelihood of an individual coming in contact with a given dose level.

(4) *Risk Characterization*. This step combines dose-response probabilities with exposure probabilities to give an overall measure of the likelihood of illness.

The goal of any health risk analysis is to quantify the likelihood of specific health problems. This helps facilitate the selection of appropriate mitigation measures to reduce risks to an acceptable level. From our hydrological and microbiological analyses we determined that there was a correlation between rain events and pollutant loading at the beach. We also found that samples containing sediments may harbor a higher concentration of indicator organisms. This section attempts to answer the question: What are the possible health risks at Arroyo Quemada and Arroyo Burro beaches based on seasonal, local indicator organism concentrations and three different dose-response relationships found in the epidemiological literature?

In this section of the project we analyzed the potential health risks associated with swimming at the two study site beaches. We used Santa Barbara County's indicator organism concentration data in conjunction with three dose-response relationships to determine a range of potential illness rates due to swimming at Arroyo Burro beach during different seasonal periods between 1996-1998 (Santa Barbara County Environmental Health & Safety Oct. 1996-Apr.1998) In addition, we used available data to calculate a range of potential illness rates for Arroyo Quemada beach at any time during the year. The explicit goals of this section are twofold:

(1) To examine how seasonal variability in pollutant loading can result in seasonal variability of potential swimming-related health risks, and

(2) To show how differences in dose-response relationships found in existing epidemiological data can effect health risk calculations, and what implications this may have in choosing appropriate public health policies.

4.1 Epidemiological Background

Historical accounts of swimming-related epidemics have long been reported. These reports illustrate the transmission of illnesses ranging from gastroenteritis, acute febrile respiratory illness, and ear, eye and skin irritation through contact with waters contaminated with domestic sewage. Over the past fifty years there has been a concerted effort to characterize the health risks associated with swimming in waters polluted with microbiological contaminants. The World Health Organization (WHO)

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and the U.S. EPA have been involved in several epidemiological studies aimed at determining the dose-response relationship for humans swimming in varying levels of an assortment of indicator organisms (Cabelli *et al.* 1982; Fleisher *et al.* 1993). However, there is still considerable disagreement as to the true relationship between various water quality measurements and illness rates.

Epidemiological studies of swimming related illness generally are of three experimental types: retrospective cohort, prospective cohort, and randomized controlled trial. Retrospective cohort studies are rarely used due to inherent inaccuracies in estimating exposure to water quality (Pruss 1998). Prospective cohort studies are the most common of the studies. In prospective cohort studies, individuals are interviewed at the beach that is being sampled for indicator organisms. The initial interview serves to determine each participant's gender, age, socioeconomic level, and previous family illness history. A follow-up interview is performed 8-10 days after the initial beach interview. Participants in the study are asked a series of questions about whether or not they had shown any specific symptoms following their day at the beach.

Of the three types of studies, the randomized controlled trial is the most accurate method with regard to matching specific water quality measurements to specific individuals. In these types of studies treatment groups that are exposed to specific levels of indicator organisms as well as a control (non-swimmer) group are selected randomly from a population of willing participants. The advantage of the randomized controlled trial is that water quality measurements can be closely linked to each interviewee. Each swimmer swims for a period of 15 minutes and a water sample is taken at chest depth at the swimmer's location. However, due to the high costs and ethical difficulties associated with such studies, only two of significance have been performed (Kay *et al.* 1994; Fleisher *et al.* 1996).

The development of current EPA contact recreational (REC-1) water quality standards for indicator organisms has been based on results from an epidemiological study conducted in the 1970s at three U.S. locations (Cabelli *et al.* 1982). This study was of the prospective cohort type, and post-exposure interview data revealed that gastroenteritis was the most common illness associated with swimming in waters containing high levels of indicator organisms. Cabelli divided reported gastrointestinal symptoms into two categories. The first category, total gastroenteric symptoms, included diarrhea, vomiting, stomachache and nausea. The second category, labeled as highly credible gastroenteric illness (HCGI), consisted of any of the total gastroenteric symptoms accompanied by a fever. It is believed that HCGI is a more accurate indicator of swimming-related illness, as other gastroenteric symptoms may arise from eating food carrying pathogenic organisms. Fleisher and Kay's studies, which were conducted in the UK, refer to total gastroenteric symptoms as subjective symptoms, and HCGI as objective symptoms (Kay *et al.* 1994; Fleisher *et al.* 1996). For the purpose of this study, we focused on HCGI, or objective symptoms, and referred to these symptoms simply as gastroenteritis.

The Cabelli study (1982) examined 10 different indicator organisms and found the strongest correlation between enterococcus levels and rates of gastroenteritis. Moreover, in a review of over 22 selected epidemiological studies, Pruss (1998) found that for marine waters, the indicator organisms best correlated with health effects were enterococci/fecal streptococci. The two UK randomized controlled studies that were included in Pruss' review found similar results. Both of these studies found that fecal streptococcus had the strongest correlation with incidence of gastroenteritis in marine waters.

Our analysis of the potential risks to human health due to swimming at Arroyo Burro beach and Arroyo Quemada beach combined Santa Barbara County sampling data with three different dose-response relationships (Cabelli *et al.* 1982; Fleisher *et al.* 1993; Kay *et al.* 1994) in order to determine average illness rates at the two study sites. The EPA dose-response relationship (Cabelli) measured illness rates in relation to enterococcus levels, while the other two studies measured illness rates in relation to fecal streptococcus levels. Santa Barbara County sampling data consisted of total coliform, fecal coliform, and enterococcus measurements. Due to the lack of species-specific data, any risk calculations made assumed that enterococcus measurements made by the County were accurate representations of fecal streptococcus concentrations. As mentioned previously in the report, enterococcus is the umbrella group that includes fecal streptococcus.

4.2 Methods

4.2.1 Introduction to Swimming-Related Dose-Response Relationships

Virtually all studies present their results in two forms. First, illness rates (or proportions) per 100 or 1000 individuals are usually reported for both swimmer and non-swimmer groups. Second, logistic regression analyses are usually performed to determine the "relative risk" between one exposure group and another. This relative risk value is reported as an odds-ratio of the two groups that are compared, with 95% confidence intervals for this ratio. Both of the UK randomized controlled studies reported higher incidence rates than did previous studies, including the Cabelli study (Pruss 1998). It has been suggested that this may be due to the fact that previous studies have had difficulty in accurately matching specific water quality to individual swimmers. However, it should be noted that the UK studies had considerably lower sample sizes than did other studies (~1000 vs. ~3000-11000) due to the high cost of conducting these studies. Furthermore, various studies have presented different illness rates for different ranges of indicator counts. Because the EPA and the UK studies each presented their dose-response relationships differently, we analyzed them using separate methodologies.

The two UK studies both presented illness rates as proportions associated with streptococcus levels and divided them into 20 MPN intervals, where MPN refers to

the most probable number of bacteria per 100 ml sample. Hence, both of these studies were analyzed using the same method. Data from these studies as well as the Cabelli (1982) study were used in conjunction with local enterococcus data to conduct a Monte Carlo simulation to determine average illness rates during different seasonal periods at Arroyo Burro beach and year round at Arroyo Quemada Beach.

4.2.2 Randomized Controlled Trial Studies' Dose-Response Relationships

We first determined the probability of the frequency of illness using the data in the two randomized controlled studies that were conducted in the UK. Based on the number of individuals examined, we determined the spread around the mean illness proportion reported in the UK study results. This was done for each treatment group (i.e. each treatment group was exposed to different levels of fecal streptococcus in 20 MPN intervals). In order to make an approximation for the 95% confidence intervals (C.I.) of each proportion for each exposure level, we used the following equation for fitting a normal distribution around a proportion, P, with sample size N:

$$95\%\text{C.I.} \quad = P \pm 1.96 \sqrt{P(1 - P)/N}$$

(Equation 4.1)

Where,
P= proportion of the population showing gastroenteric symptoms
N= number of individuals in the treatment group
(Wonnacott and Wonnacott 1990)

Equation 4.1 provides a good approximation of the sampling distribution of the means for large sample sizes. The definition of a large sample size is when at least 5 successes and 5 failures can occur within the same sample size. In other words, N should be at least 10 (Wonnacott and Wonnacott 1990). For the most part, the sample sizes we dealt with were on the order of $N=20$ to $N=608$. Typically, for both studies, fewer individuals were exposed to high indicator concentrations, while there were a greater number of individuals in the non-swimmer and low concentration treatment groups. Using this approximation to determine the probability density function of the proportions, we obtained an estimate of the spread about the mean for each treatment level.

For both studies, we then determined the difference between illness rates of the non-swimmer and swimmer groups, and the corresponding 95% confidence intervals associated with this difference.

$$(P_{swimmer} - P_{non-swimmer}) \pm 1.96 \sqrt{[(P_s(1-P_s)/N_s) + (P_{ns}(1-P_{ns})/N_{ns})]}$$

(Equation 4.2)

Where,

P_s = proportion of swimmer group population reporting gastroenteritis
P_{ns} = proportion of non-swimmer group population reporting gastroenteritis
$N_{s/ns}$ = number of individuals in each treatment group
(Equation adapted from Wonnacott and Wonnacott 1990)

Equation 4.2 gives swimming-related illness rates for every exposure group by subtracting the background non-swimmer illness rates from swimmer group illness rates (Kay *et al.* 1994; Fleisher *et al.* 1996). Both studies have performed chi-squared analyses between the swimmer and non-swimmer groups to determine if there were significant differences between the various exposure groups and the non-swimmer group. Both came to the same conclusion that there was no significant difference in rates of gastroenteritis between swimmers exposed to bacterial concentrations ranging from 0-39 MPN of fecal streptococcus and the non-swimmer groups. Hence, according to these studies, the threshold level of fecal streptococcus that causes excess illness to the population at large is 40 MPN. Appendix XI shows a plot of the data from these two studies, with the non-swimmer rates subtracted from the swimmer illness rates. If the non-swimmer rate was higher than the swimmer rate for a given exposure group, then a value of zero excess illness was attributed to treatment groups exposed to bacteria concentrations less than 40 MPN.

For each exposure level, we obtained different normal curves describing the spread about the mean (Appendix XII). The larger the sample size, the more certain we were about having the sample population represent the real population response. For lower sample sizes, the confidence intervals were broader. This is due to the fact that small sample sizes are not good representations of the long-range frequency of illness. Once we determined the spread of the normal distribution for each exposure level, the discrete dose-response relationships were used in a Monte Carlo simulation in conjunction with a probability mass function to describe the frequency of indicator organism loading levels for Arroyo Burro and Arroyo Quemada.

4.2.3   EPA Dose-Response Relationship

We employed another method for determining the average illness rate for a region based on the EPA dose-response function developed in the Cabelli study (Cabelli *et*

al. 1983). The EPA dose-response relationship described illness rate as a logged function of enterococcus concentration:

$$R(C) = .20 + 12.17 \log(C) \qquad \text{(Equation 4.3)}$$

R is the rate of gastroenteritis per 1000 individuals and *C* is the enterococcus concentration in MPN. This relationship gives significantly lower illness rates than the relationships derived from the UK studies. For example, the illness rate at doses around 80 MPN calculated from the EPA relationship was on the order of 23 individuals reporting gastroenteric symptoms per 1000 individuals, as opposed to the rate of 100 per 1000 individuals obtained from the UK studies. This is a disparity of nearly one order of magnitude. Moreover, the EPA study did not divide its treatment groups into equal concentration levels. Hence, it was not possible to determine the spread around the mean for distinct concentration levels for the EPA dose-response relationship. The EPA dose-response function is a continuous function that attempts to model the behavior of the data collected in Cabelli's study. Unfortunately, the EPA relationship does not consider the uncertainty or population statistics associated with illness rates. It merely provides a measurement of the mean illness rate at any given concentration without accounting for standard deviations or confidence intervals associated with the mean.

4.2.4   Indicator Organism Loading Concentrations

Elliott (1998) showed how the EPA dose-response relationship can be used in conjunction with a log-normal bacteria loading frequency distribution to determine average expected rates of illness. The assumption of a log-normal loading frequency distribution may not be valid for most regions. These distributions may be directly linked to rain/flushing events. In this case, a distribution where low levels are predominant during dry periods (and high levels are predominant during wet weather conditions) would characterize the frequency of loading levels more appropriately. In order to account for seasonal variability in indicator counts at the beaches in question, we separated County enterococcus measurements at Arroyo Burro into wet and dry seasonal categories. Chronically high enterococcus measurements at Arroyo Quemada beach showed no seasonal variability; therefore, only a single loading graph was created that combined all available data for this beach. Appendix XIII shows the 20 MPN interval probability mass graphs of the frequency of enterococcus loading at Arroyo Burro Beach and Arroyo Quemada beach for sampling dates in 1996-1998. The probability mass graphs show the probability of occurrence of a 20 MPN enterococcus interval based on historical data. In these graphs, enterococcus levels were divided into discrete 20 MPN intervals, up to 80+ MPN. This allowed for direct combination of the loading probabilities with the dose-response relationship found in the two UK studies.

Monthly precipitation data for each watershed were analyzed to determine seasonal precipitation trends for 1997 and 1998 (Table 4.1). Then estimates were made based

on these precipitation trends to determine what the effective "rainy" and "dry" seasons were for these two years. These seasonal periods were then used to divide the enterococcus data for the two-year period into wet and dry season categories for Arroyo Burro. Again, due to chronically high bacterial levels at Arroyo Quemada, bacteria levels were not divided into wet and dry seasons. The probability mass graph for Arroyo Quemada simply shows the breakdown of enterococcus levels for 66 sampling dates into the 20 MPN intervals. The ocean indicator organism data used to create these graphs was the same data used in the microbiology section of our report (Santa Barbara County Environmental Health and Safety 10/96-5/98). These ocean water samples were taken at the surf zone, directly in front of creek mouth, weekly. There were no available data to show how indicator organism levels varied with distance from the creek mouth. Hence, our analysis focused on health risks associated with swimming in ocean water directly adjacent to creek outfalls.

Table 4.1 Summary of Seasonal Precipitation Data for Arroyo Burro

Seasonal Period	Dates	Total Rainfall (in.)
Wet '96/'97	10/31/96 - 1/27/97	19.47"
Dry '97	2/3/97 - 11/3/97	0.48"
Wet '97/98	11/10/97 - 6/1/98	70.34"
Dry '98	6/8/98 – 10/5/98	0.35"

In order to determine more accurately average illness rate based on the EPA relationship, we split the loading data at each beach into as many discrete enterococcus levels as possible. Appendix XIII shows the discrete probability mass graphs used in conjunction with the EPA dose-response curve for both beaches. Again, the loading probabilities for Arroyo Burro are divided into wet and dry seasons, and only one loading graph was made for Arroyo Quemada.

4.2.5   Monte Carlo Simulation

Once the probability mass functions and dose-response relationships were determined, they were combined using a Monte Carlo simulation to calculate mean illness rates at each beach. The Monte Carlo simulation was performed using @RISK (product of Palisade Corp., Newfield, NY) add-in for Microsoft Excel. The simulation was run for 2000 iterations so as to reduce the variance in the final outcome. For each simulation iteration, @RISK selected an enterococcus loading concentration from each probability mass function graph. Loading levels with higher probabilities were selected more frequently than loading levels with lower probability values. For each iteration, the randomized controlled trial illness rates were calculated using a series of embedded logical statements that instructed the spreadsheet to match up a selected probability loading interval (i.e. 0-19, 20-39, etc.) from Appendices XIII (figures b and c) with the appropriate dose response curve in Appendix XII. For both randomized controlled dose-response relationships, an illness rate of zero was attributed to any loading level below 40 MPN. This was done to account for the

statistical insignificance between the non-swimmer and swimmer groups at these lower indicator organism levels. Results were compiled for all 2000 iterations, and all the statistics were calculated using @RISK.

Illness rates using the EPA dose-response function were calculated in the following way. For each iteration one enterococcus level was selected from the discrete probability mass functions in Appendix XIII. The selected enterococcus levels were then entered into equation 4.3 to determine the proportion of the population that could potentially contract gastroenteritis. This simulation was also run for 2000 iterations, and all statistics for the iteration outputs (illness rates) were calculated by @RISK.

4.3   Results

Table 4.2 shows the calculated mean illness rates and standard deviations for each seasonal period at Arroyo Burro, and for the entire sampling period at Arroyo Quemada. The mean illness rate, μ, is the fraction of the population expected to contract gastroenteritis. The standard deviation, σ, is the standard deviation of the calculated mean illness rate from the 2000 iterations of the risk model. A major artifact in the presented data is that the standard deviations for the EPA derived illness rates are significantly lower than those found for the other two studies. This is due to the fact that the EPA dose-response relationship did not take into consideration the uncertainty associated with a calculated illness rate for a given enterococcus level.

Appendix XIV shows the means, standard deviations, and 95% confidence intervals of the calculated illness rates. At a glance we see that the illness rates calculated using the UK studies tend to be higher than those calculated using the EPA dose-response relationship. As will be discussed later, this disparity was not as prevalent for the more highly contaminated Arroyo Quemada beach site. Table 4.2 clearly shows that the predicted illness rates for Arroyo Burro beach were higher during the wet seasonal periods than in the dry seasonal periods. These results are also summarized in Appendix XIV (figures a and b). Moreover, further analysis of the results show that illness rates were generally higher for Arroyo Quemada than Arroyo Burro even during wet seasonal periods of high loading at Arroyo Burro.

Table 4.2 Calculated illness rates for Arroyo Burro and Arroyo Quemada

Season/Site	Fleisher *et al.* (1993)	Kay *et al.* (1994)	EPA-Cabelli (1983)
Wet '96-'97/Arroyo Burro	μ=.1613 σ=.1354	μ=.1347 σ=.1079	μ=.0244 σ=.008955
Dry '97/ Arroyo Burro	μ=.0484 σ=.1028	μ=.03712 σ=.008051	μ=.01133 σ=.001011
Wet '97-'98/ Arroyo Burro	μ=.1981 σ=.1409	μ=.1571 σ=.1136	μ=.02576 σ=.009953
Dry '98/ Arroyo Burro	μ=.1090 σ=.1411	μ=.08515 σ=.1103	μ=.01841 σ=.008830
All Sampling Dates/ Arroyo Quemada	μ=.2614 σ=.1217	μ=.2050 σ=.09896	μ=.2357 σ=.007042

4.4   Discussion

There is considerable disparity between the illness rates calculated using the UK studies' relationships and those calculated using the EPA relationship. First, higher illness rates were found by both UK studies. Pruss (1998) suggests that this may be due to the fact that the UK studies more accurately matched water quality measurements to individual swimmers. In our analysis, the disparity between the UK and EPA relationship becomes less during periods of high indicator bacterial loading. This phenomenon is due to the fact that a continuous relationship for illness rate was not provided by the UK studies as it was in the EPA dose-response function. All enterococcus levels above 80 MPN gave the same illness rate when using the UK relationships in our analysis. Conversely, the continuous EPA dose-response relationship generated incrementally higher illness rates at higher enterococcus levels. This disparity has two salient implications. First, there is the need for all epidemiological studies to present their results in a uniform and useable form so as to allow others to consistently compare results. Second, it may be useful to combine expert data from many epidemiological studies to determine a consensus dose-response function that incorporates uncertainty based on all available data. In Pruss' analysis of 22 epidemiological studies, she laid out the groundwork for such a curve to be formulated. However, one consensus dose-response curve may not be an accurate measure of human response, due to several factors. First, there may be regional differences in human illness response due to various physiological factors and immunities of different populations. Second, waters in various regions of the world may have diverse pathogen assemblages that may be masked or overstated by simple indicator organism measurements (Fleisher 1993). Finally, age may be a significant factor in determining who is susceptible to contracting illnesses.

Our comparison also shows the need for uniform methodologies when conducting epidemiological studies. The two randomized controlled studies show how matching water quality measurements directly to individual swimmers can greatly affect observed illness rates. In the epidemiological literature there is an ongoing dispute on what the actual dose-response curve should be. Fleisher (1993) performed a reanalysis of Cabelli's data and found key methodological weaknesses in the way the data was analyzed. First, he notes that "results obtained from brackish water locations were pooled in the final analysis of the data." Evidence suggests that indicator organisms survive longer in waters with lower salinity, and this may result in different ratios of indicator organisms to underlying pathogens (Dufour 1984). Fleisher claims that this pooling of the data greatly affected the resulting dose-response function. Fleisher computed three separate dose-response curves based on the separation of the brackish water data and the marine data. This reanalysis found distinct differences between the dose-response relationships at each of the three study sites. It is apparent that there are many factors that can confound the resulting dose-response curve. Previous attempts to reduce the number of confounding factors include matching water quality to individual bathers, using uniform illness criteria, considering family history and food intake in the interview process, and using similar water quality measurements. However, there is still need for uniformity in the way studies are conducted.

The results suggest a trend of higher risk of contracting gastroenteritis during periods of increased rain. Table 4.2 shows that calculated illness rate levels were higher during both wet seasonal periods when compared to corresponding dry seasonal levels. As suggested in our sediment study, it is reasonable to expect increased levels of indicator organisms in creek water and at the ocean receptor site after periods of high creek flow. This is due to the mobilization and transport of organisms residing on sediment particles through the riparian environment. This result has important implications for public health notifications. The public should be well informed that during periods of high creek flow, individuals who choose to swim in marine waters should attempt to keep away from creek outflows. This recommendation is further corroborated by the Santa Monica Bay Restoration Project's epidemiological study (Haile *et al.* 1996). In this study it was determined that swimmers exhibited a higher risk of contracting gastroenteritis, as well as other illnesses, when exposed to marine waters near creek mouths.

Appendix XIV shows that there is considerable variance in the UK calculated illness rates. This is primarily due to the fact that the sample sizes for some of the treatment groups were fairly small. When conducting a randomized controlled study it is difficult to have large numbers of willing participants and the protocol may be very costly and time consuming. There is a trade-off between having water quality measurements matched with individual bathers, and having large numbers of participants. If the total sample sizes were as large as 10,000 individuals, our analysis would have shown a smaller 95% confidence range for each calculated illness rate.

It should also be noted that the health risks for the dry 1998 seasonal period were considerably higher than those for the dry 1996-1997 period. There was relatively higher loading at Arroyo Burro beach during the dry 1998 period in contrast to the dry 1996 period. This is presumably due to the high volume of creek flow associated with the intense El Niño rains of the previous winter and spring. Due to limited data, the separation of the data into seasonal loading periods was based on precipitation events rather than creek flow. The increased loading during this dry period could be attributed to the relatively large volume of groundwater discharge into the creek that year. A simulation of groundwater discharge and overland flow could have provided a more precise way of splitting the data into low flow versus high flow regimes. Nevertheless, our method provides a good approximation of the loading probabilities during the different flow regimes.

The nature of the uncertainty in using indicator organisms to measure risk begs the question of whether or not other types of organisms should be sampled. Cabelli (1982) mentions that human rotaviruses or Norwalk-like viruses might be the key pathogenic agent causing gastroenteritis. Furthermore, the Santa Monica Bay Restoration Project (SMBRP) study found that several of the deleterious health effects were reported more often on days when samples tested positive for viral agents (Haile *et al.* 1996). Wyer *et al.* (1995a) found that "the amount of variance in enterovirus concentration explained by bacterial concentration was only 15-16%." If enteroviruses, rotaviruses, adenoviruses, and Norwalk-like viruses are a significant cause of swimming related illness then this could explain why indicator organisms do not accurately measure risk. However, the currently used indicator tests do have many advantages in that they are cheaper and easier to perform than virus tests, and they do account for a variety of potential pathogens in genera such as *Echerichia*, *Klebsiella*, *Vibrio*, etc. Unfortunately, as we explained earlier, the use of total coliform, fecal coliform, and enterococcus may give false positive results in terms of measuring organisms not of human fecal origin. Until more advanced pathogen identification technologies become available, the current cost-effective sampling methods seem to be most appropriate.

In this report, we focus only on the correlation between enterococcus and gastroenteritis. Other studies have attempted to find links between indicator counts and non-enteric illness. Fleisher *et al.* (1996) found that fecal streptococci were predictive of acute febrile respiratory illness and fecal coliform were predictive of ear infections. However, they found no correlation between indicator levels and eye infections. In addition, the SMBRP study found an increased incidence of skin rashes for those individuals swimming in high levels of fecal coliform. These findings would suggest that the use of one single indicator organism, such as enterococcus, may not be adequate to monitor all potential illnesses, and using multiple standards is appropriate. Moreover, future investigations similar to ours may wish to incorporate data on non-enteric illnesses into their health risk analyses.

4.5   Future Directions, Conclusions, and Management Implications

4.5.1   Future Directions- Cost of Illness

Of considerable interest is the severity of swimming-related illnesses. As part of the two UK studies, individual physical examinations were conducted for all participants in the studies (Fleisher *et al.* 1998). The researchers found that the length of illness tended to be longer for those in the swimmer groups than those in the non-swimmer groups. Participants were examined for four illnesses: gastroenteritis, acute febrile respiratory illness, ear infections, and eye infections. In each illness category, except for acute febrile respiratory illness, bather groups had lengthier illness periods. In addition to measuring severity of illness, they also determined what percentage of the test population sought medical treatment (4.2%-22.2%), as well as what percentage lost at least one day of normal activity (7.0%-25.9%). Based on this analysis they also determined that the "overall percentage of each illness directly attributable to exposure to marine waters contaminated with domestic sewage ranged from a low of 34.5% for gastroenteritis to a high of 65.8% for ear infections." These findings suggest that there may be considerable economic impacts associated with the growing problem of water polluted by domestic sewage. Further studies may choose to use these findings along with a method similar to ours of determining average illness rates, in order to determine the potential economic impacts bacterial contamination. Such an economic analysis of the cost of illness associated with swimming in marine recreational waters could be used in a cost-benefit analysis to help state and regional planners formulate pollution control policies. A framework for understanding the physical, biological, and economic factors involved would provide a tool for policy makers to make well-informed, cost-effective decisions.

4.5.2   Future directions- System analysis and quantified logic trees

Logic trees can be extremely useful in the design phase of analyzing environmental risks. Due to the fact that natural systems have a tremendous amount of variability and outside influences, it becomes necessary to break up these complex systems into a series of more digestible pieces. These pieces can then be combined to create a bigger picture of the system as a whole. Each individual system component or process lends itself to event tree analysis. By combining knowledge about the mechanistic (e.g. chemical, physical, biological) processes of each system component with some known data about how the system has worked in the past, a logic tree can be created that attempts to characterize individual components of the system, and the probabilities associated with specific outcomes.

Understanding the complex dynamics of bacterial contaminant origination and transfer dynamics throughout watersheds requires a great deal of knowledge from many disciplines. Logic trees can be designed to include mechanistic modeling approaches to describe various processes that result in the release and transport of biological contaminants to the environment. These types of logic trees can be used to

incorporate knowledge of physics and engineering to characterize the failure of septic systems and sewage lines under various environmental stresses such as temperature, pressure, and moisture changes. Furthermore, knowledge about how hydrological and biological parameters influence the macro-scale and micro-scale transport of these pollutants both via groundwater transport and surface flow transport mechanisms can be incorporated into such logic trees. If we consider bacteria and viruses as simple colloids, then modeling these processes can be fairly straightforward with the use of pollutant loading models. Unfortunately, biological agents do not behave this way, and they are subject to many different stresses in the environment that can both enhance and hinder their growth and persistence in the environment. Some research is currently being conducted to model the movement of enteric bacteria through soil media (McMurray 1998).

Several studies have been conducted that have developed models for bacterial transport, and survival/death rates based on mass balance loading and loss rates within watersheds and in lakes (Auer *et al.* 1993; Canale *et al.* 1993). These models include variables such as wind speed, death by UV irradiance, loss due to sedimentation, and dispersion coefficients. These complex models have been formulated separately from health risk studies, and there is a need for the integration of this type of mechanistic knowledge into a comprehensive risk analysis framework. The main advantage to using a probability-based framework to analyze this pollution problem is that pollutant loads can be linked to specific system component failures. Scientific knowledge can be used to determine how likely a pollutant is to be released from a specific source, and the likelihood of pathogens reaching an area where humans are more likely to come in contact with the pollutants. Moreover, a systemic risk model could greatly be enhanced if indicator organisms could be linked to specific types of system failures. For example, if the genetic fingerprinting analysis (Samadpour 1995) described earlier could be used to accurately pinpoint pollution sources, then this type of data could aid in the calibration of a systemic risk model that incorporates spatial and temporal pathogen transport mechanisms. This type of systemic approach could be used to select the most cost effective mitigation solutions based on available scientific data.

4.5.3   Conclusions and Management Implications

The risks associated with swimming in contaminated marine recreational waters can be classified as being relatively high frequency, mid to low consequence risks on the risk continuum scale. Nevertheless, the public outcry in Santa Barbara makes this a problem worthy of attention. Individuals relatively measure health risks. One individual's perceived risk may be quite different than another's. For instance, in the eyes of a decision-maker in a poor region, high indicator bacteria levels may be of little importance due to a relatively uninformed public and a lack of public resources to do anything about it. However, in a wealthy, environmentally conscious community such as Santa Barbara, the public demands that expeditious governmental action be taken to reduce these mid to low consequence risks.

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The goals of analyzing health risks are to understand the actual risks associated with the pollution problem, and to be able to take appropriate action to reduce the incidence rate in the exposed population to some acceptable level. This acceptable level is largely dependent on socioeconomic factors. Also, as our analysis shows, an acceptable risk level is dependent upon the dose-response relationship used to quantify the potential health risks at a particular swimming location. It is the responsibility of the state and local agencies to implement pollution control measures to reduce the amount of coastal pollution at least to the federally mandated level. If a locality so chooses, it may set standards stricter than those required by federal statutes such as the Clean Water Act.

5.0   Legal Enforcement Mechanisms

Since legislation is such a persuasive, if not unwieldy, tool in the effort to reduce coastal bacterial contamination, it is useful to examine the relevant federal and state laws and local ordinances that apply to point and non-point source pollution. Legislation such as the Clean Water Act defines the parameters of the effort required by Santa Barbara to comply with legal mandates to clean up coastal pollution. In addition, statutory requirements coupled with court decisions often determine the sufficiency of scientific efforts designed to answer difficult questions. Environmental groups unhappy with agency determinations often seek more favorable clarifications from judges. It is the intent of the members of this project to provide management as well as scientific recommendations that may serve as signposts to successfully improve water quality under the statutory requirements of the Clean Water Act and other legislation.

5.1   The Clean Water Act

The Clean Water Act (CWA) was passed in 1972 to "restore and maintain the chemical, physical and biological integrity of the nation's water" (US Congress 1998a). Until the 1960's, water pollution control was based on the principles of nuisance and the balance of competing uses by local water boards and in a few instances local courts (Houck 1997). In 1965, Congress passed the Water Quality Act that requires federally approved standards on interstate waters. These water quality standards were not effective. "The Senate found the standards weak, late, widely disparate, scientifically doubtful, largely unenforced, and probably unenforceable" (Houck 1997). During the congressional hearings for the passage of the CWA, the Acting Chair of the House Public Works Committee stated the following regarding state efforts for water pollution clean up: "We left it to the States, year after year, and we didn't get a single thing but a bunch of nursery rhymes as the Constitution, and we didn't get any clean water until the Federal Government insisted upon it and made some dollars available to the State for that use" (US Congress 1971b).

When the CWA was written, Congress focused on two sources of pollution: point and nonpoint source pollution. A point source is defined under the CWA as "any discernible, confined, and discrete conveyance," including pipes, ditches, conduits, or vessels "from which pollutants are or may be discharged" (US Congress 1998c). Scrapping the water quality standards approach of the 1965 Act, Congress adopted a technology-based approach for the abatement of point source pollution, and required point sources to obtain a National Pollution Discharge Elimination System (NPDES) permit. NPDES permits established numerical limits on the amounts of pollutants released from each point source and adopted a technology based approach to reduce pollution from point sources such as wastewater treatment plants. It should be noted that the NPDES program has been highly successful in reducing pollutants, such as coliform in the water, from being discharged from treatment plants (Higgins 1999).

Congressional legislation under the Clean Water Act next tackled pollution from storm drains, which had not been dealt with in 1972. In 1987, the law was amended to require NPDES permits for storm drains in areas with populations greater than 100,000. Phase II of the storm drain regulation was published in the proposed rules in the *Federal Register* on January 9, 1998, requiring NPDES permits for communities under 100,000. Arroyo Burro falls within this requirement, and the recent sampling under Project Clean Water and the South Coast Watershed Characterization Study is a direct result of the new regulations. Under the permit, the county is required to include various Best Management Practices (BMPs). BMPs are defined by federal regulations as "schedules of activities, prohibitions of practices, maintenance procedures, and other management practices to prevent or reduce the pollution of waters of the United States" (USEPA 1998a). BMPs also include treatment requirements, operating procedures, and practices to control plant site runoff, spillage or leaks, sludge or waste disposal, or drainage from raw material storage (USEPA 1998a). In addition to BMPs, the regulations also mandate the development of Total Maximum Daily Loads (TMDLs).

5.2   Total Maximum Daily Loads

The second type of pollution dealt with by the CWA was runoff pollution, or nonpoint source pollution, defined as any non-discrete source, such as runoff from agriculture, forestry, or construction activity. In 1972, Congress adopted a water quality standard to address runoff pollution. The mechanism created was Total Daily Maximum Loads (TMDLs). TMDLs are defined as: "the sum of the individual wasteload allocations for existing and future point sources (including storm water) and load allocations for existing and future nonpoint sources (including diffuse runoff and agricultural storm water) and natural background materials with a margin of safety incorporated to account for uncertainty in the analysis" (US January 9, 1998). Congress intended that a water quality standard be applied to waters that remained polluted after the development and application of technology-based standards.

Under the TMDL process, a state must adopt water quality standards based on the uses of the waters and the amount of pollution that would impair the uses. Then, waters in states that do not meet these water quality standards are identified. These are known as "water quality limiting segments" (WQLS). After these WQLSs are identified, they are prioritized according to the severity of the pollution and the uses of the waters. Finally, TMDLs are developed for each pollutant impairing each WQLS according to the priority ranking.

A TMDL sets the maximum amount of pollutants a water body can receive daily without violating the state's water quality standards (US Congress 1998b). A TMDL includes best estimates of pollution from non-point sources and natural background sources known as load allocations (LAs), pollution from point sources known as wasteload allocations (WLAs), and a margin of safety (USEPA 1998b). TMDLs must also take into account seasonal variations (US Congress 1998b).

Congress wrote section 303(d) of the CWA in response to the failings of the Water Quality Act of 1965 (Houck 1997). Under section 303(d) of the CWA, states were required to take the following steps:

1.   Identify waters that are and will remain polluted after the application of technology standards.
2.   Prioritize these waters, taking into account the severity of their pollution.
3.   Establish "total maximum daily loads" for these waters at levels necessary to meet applicable water quality standards, accounting for seasonal variations and with a margin of safety to reflect lack of certainty about discharges and water quality (US Congress 1998b).

The list of polluted waterways is referred to as the "303(d) list." According to one congressional staff member at the time, "We didn't take it [303(d)] seriously and thought it would be foolish for EPA to waste time and money to implement it" (Houck 1997). Today, the TMDL program is "the centerpiece of the president's Clean Water Action Plan," according to EPA Administrator Carol Browner (*The Environmental Forum*, p. 36).

However, the entire TMDL process lay dormant like Mt. Saint Helens for many years because of the reluctance of policymakers to undertake a difficult procedure. That is, it took lawsuits by citizen's groups under the CWA to force the EPA and the states to begin development of TMDLs. For example, Heal the Bay recently reached a settlement in a lawsuit for the development of TMDLs for Los Angeles and Ventura counties (*Environmental Science and Technology News* 1999). In *Sierra Club v. Hankinson*, a federal judge in 1996 ordered the state of Georgia to identify every polluted waterway in the state and set TMDLs for these waterways, and examine the entire watershed if necessary. The development of TMDLs for Arroyo Burro is just beginning.

5.3   Management of Coastal Pollution at Arroyo Burro

> There is no effective way as yet, other than land use control, by which you
> can intercept that runoff and control it in the way that you do a point source.
> We have not yet developed technology to deal with that kind of a problem.
> We need to find ways to deal with it, because a great quantity of pollutants is
> discharged by runoff, not only from agriculture but from construction sites,
> from streets, from parking lots, and so on, and we have to be concerned with
> developing controls for them (US Congress Nov. 2, 1971).

There are numerous state laws for the abatement of coastal pollution at Arroyo Burro.
For example, discharge of sewage or other waste that results in contamination,
pollution or a nuisance is prohibited under section 5411 of the California Health and
Safety Code. However, one of the difficulties in enforcing pollution control of runoff
pollution is identifying the source of pollution. This requires sampling of numerous
sites and with a frequency sufficient to foreclose uncertainty. It was not until the
passage of AB 411 that health officials were required even to formally notify the
public of a contaminated beach. However, notification is only the first step towards
solving the problem of contaminated beaches (California State Assembly 1998).

As part of the county's effort towards applying for an NPDES phase II storm drain
process permit, water samples have been taken at Arroyo Burro beach and upstream
locations. Under the NPDES permit, the county is also required to institute various
BMPs "designed to reduce the discharge of pollutants from your municipal separate
storm sewer system to the maximum extent practicable (MEP) and protect water
quality"(US January 9, 1998). At a minimum, the regulations require the following
BMPs:

1. Public education and outreach on storm water impacts. Examples of this
   BMP can be seen in the Santa Barbara weekly newspaper, *The
   Independent* and in other newspapers as part of the "Save Our Shoreline"
   campaign;
2. Public involvement/participation, such as public hearings;
3. Detection and elimination of illicit discharges into the storm sewer
   system;
4. Development of a program to reduce pollutants in storm water runoff
   from construction activities involving one acre or more;
5. Development of a program to reduce pollutants from new development
   and redevelopment projects of one acre or more;
6. Development of a program to prevent or reduce runoff from municipal
   operations.

It should be noted that this is not an all-inclusive list. There are other BMPs available,
including land use restrictions, zoning, and retention ponds. Zoning and land use
restrictions are subject to approval by the Santa Barbara Board of Supervisors and are

not statutorily mandated under the Clean Water Act. Approval of zoning amendments is subject to political factors as well as a possible constitutional issue of taking. The issue of taking is complex, and requires an analysis of several factors, including the "nexus" of the land use restriction to the governmental purpose. For example, a height restriction ordinance for beach front property has no nexus to a public access to the beach front ordinance. However, a land use restriction to reduce runoff from non-porous land cover may very well have sufficient nexus to pass legal scrutiny. This is an issue that the county will have to examine carefully.

The proposed regulations recommend that the storm water permit include a monitoring program to gather necessary information in order to determine the extent of attaining water quality standards and to determine the appropriate conditions or limitations of subsequent permits. This monitoring program can include ambient monitoring, an assessment of the quality of receiving waters, discharge monitoring, or a combination. The regulations also state that if a BMP-based program fails to protect water quality, and "if specific measures to protect water quality were imposed, they would likely be the result of an assessment based on TMDLs, or the equivalent of TMDLs, where the proper allocations would be made to all contributing sources" (US January 9, 1998).

Thus, the twisted regulatory trail of the CWA leads to TMDLs.

5.4   The Role of Science

The first step in the development of TMDLs is the determination of the applicable water quality standard. The Water Quality Control Plan for Ocean Waters of California ("Ocean Plan") defines ocean water quality for regional waters as 1000 MPN/100 ml for total coliform, and 200 MPN/100 ml for fecal coliform (California State Code 1994). These standards apply to ocean water 1000 feet from shore and down to a 30-foot depth. Note that this is the same standard as AB 411. It appears that this will be the water quality standard that applies to Arroyo Burro Beach. However, there is no freshwater standard at this time for the Arroyo Burro watershed. Therefore, one issue is whether the Ocean Plan will apply to waters upstream from Arroyo Burro Beach.   The county is currently considering whether there will be a different standard upstream, which if instituted, would result in one standard of water quality upstream and another standard at the coast. It is not clear if this meets the intent of the Clean Water Act.

When the Clean Water Act was written in 1972, the intent of Congress was to eliminate "discharge of pollutants into the *navigable* waters" of the United States by 1985 (US Congress 1998a) (emphasis added). Furthermore, a Senate report at the time of the Act's conception states that water standards are expected to establish the maximum level of pollution allowable in *interstate* waters (US Congress 1971a) (emphasis added). The point here is that Congressional intent seem to focus on the

waterway in toto rather than segregate into freshwater and ocean water. However, the proposed regulations for the phase II storm drain NPDES permits state that

> Water quality standards are the cornerstone of a State's or Tribe's water quality management program. States and Tribes adopt water quality standards for waters within their jurisdictions. Water quality standards define a use for a waterbody and describe the specific water quality criteria to achieve that use. Examples of designated uses are recreation and protection of aquatic life. Water quality criteria can include chemical, physical, or biologicial parameters, expressed as either numeric limits or narrative statements. The water quality standards also contain antidegradation policies to protect existing uses and high quality water.
> (US January 9, 1998).

Thus, it is entirely within the regulatory intent to designate a water quality standard for recreational use at beaches such as Arroyo Burro Beach. If a different water quality standard exists upstream, this still leaves the issue of how the incoming waters will achieve the standard of the Ocean Plan. There is also a question of public health and welfare if freshwater standards are in excess of coastal water standards. This is an issue that Santa Barbara health officials will have to consider carefully.

Determining the appropriate water quality standard is just the tip of the iceberg in the lengthy process of developing TMDLs for Arroyo Burro. Load allocations for non-point sources as well as Waste Load Allocations (WLA) will have to be developed based on sampling data. Arroyo Burro is on the 303(d) list for pathogen pollution. It has been given a medium priority and has been scheduled for a start date of April 2006 and completion date of April 2011. Six miles of waterways have been identified as impaired and will require data collection (Mike Higgins 1999). In order to identify sources of non-point source pollution, more data will have to be collected. Various hotspots of potential pollution have been identified, but it appears that only the collection of sampling data will conclusively identify sources. If the county monitors storms drains on a regular basis under the NPDES permit, it is likely that this will build a database for the WLA for the storm drain point source (Fleishly 1999). Other possible sources such as bacteria in sediment present a challenge for monitoring and identification.

The sampling at Arroyo Burro has raised various issues of reliability and frequency of samples. In fact, "legal challenges are already rising over the degree of science necessary to support load calculations and their allocations to particular sources" (Houck 1997). The Chemical Manufacturers Association has advocated that no TMDL restrictions be instituted until the science of each assessment is validated. "The degree of supporting science is clearly where the challenges to TMDLs will lie" (Houck 1997). Calculating load allocations is the critical feature of TMDLs, and one which the Santa Barbara Health Department and the Regional Water Quality Control Board is going to face. However, "allocations of loadings to particular sources in the

TMDL process is entirely political, as it is in the analogous state implementation plan process of the Clean Air Act; the mix of reductions from point and nonpoint sources a state may choose to meet its ambient standards is a matter for the state to legislate, negotiate, or otherwise determine" (Houck 1997).

Thus, the supporting science runs up against the political process, which will have implications for the development of TMDLs for Arroyo Burro.

5.5   Numerical or Narrative?

There is also an issue of how the load allocations for point and non-point source pollution will be written. Will a narrative description suffice, or will numerical values be required? In the case of *Environmental Defense Fund, Inc. (EDF) v. Costle* (1981), the plaintiffs sued the EPA for inadequate water standards for the Colorado River. In an effort to reduce the salinity of the Colorado River from salt loading in runoff waters, the Colorado River Basin states had adopted water quality standards. The salinity was caused by river water dissolving salt from soils and rocks, with half of the salt arising from natural causes and the other half from human activities. The salinity standards included specific numeric criteria for three monitoring stations in the River's lower stem, narrative provisions, and other factual information, with the goal of maintaining salinity concentrations below 1972 levels.

EDF claimed in the lawsuit that separate numeric criteria were to be established in each basin state and that a failure to do so created a set of salinity standards with no accountability. What the court held has direct impact on the development of water quality standards for Arroyo Burro.

> Here, EDF details several reasons which it argues necessitates judicial disapproval and corrective remand of the salinity standards. EDF first contends
> that the Clean Water Act and corresponding EPA regulations provide that numeric criteria are needed in each of the seven states. To the contrary, neither the Act itself nor the regulations require that any numeric criteria be established. Water quality criteria may be, and often are, totally narrative. If the establishment of numeric criteria in each state became legally mandated after thorough EPA study and review of its statutory obligations, EPA would have been duty bound to promulgate appropriate regulations (Environmental Defense Fund, Inc. v. Costle 1981).

As a consequence, the county and the Regional Water Quality Control Board may choose to define water quality standards by narrative values rather than by specific numerical values, which is not favored by environmental groups such as Heal the Bay. A numerical value is much more precise and therefore easier to enforce.

5.6   TMDLs or BMPs?

The TMDL process is an ongoing process. Targets are established, measures are instituted, monitoring is conducted, and then the measures are reevaluated to see if targets are met. If they are not met, then more measures are instituted. Therefore, it is likely that the TMDL for Arroyo Burro will be revised in the future based on sampling.

Even though TMDLs are in their infancy, their potential to compel agencies to clean up nonpoint source pollution such as that at Arroyo Burro is much greater than BMPs. BMPs are voluntary, and voluntary measures in the past 27 years have largely been unsuccessful. However, TMDLs "are the basis of watershed planning, not because they are scientifically bulletproof, comprehensive, or efficient, but because they are objective, measurable, and the only approach so far that can be enforced by law" (Houck 1998). To what extent they will be enforced remains to be seen.

It should also be noted in the context of enforcement mechanisms that the Coastal Zone Amendments Reauthorization Act (CZARA) of 1990 requires states to identify both land use sources and coastal water quality and develop "management measures" to achieve water quality standards under the Clean Water Act. However, as of spring 1998, not one coastal state had submitted an approved nonpoint program under CZARA (Houck 1998). Like BMPs, these management measures have no teeth under the law, and thus the expectations of TMDLs are raised even higher.

TMDLs may develop into a powerful impetus to institute meaningful and effective BMPS, such as land use zoning and retention ponds. As the county of Santa Barbara and the Regional Water Quality Control Board continue to assess the coastal contamination at Arroyo Burro Beach and other popular ocean recreational spots, such as Rincon Beach, they may realize that public attention will require tough measures to improve water quality for impaired waterways. Environmental groups have already demonstrated that they are willing to challenge what they perceive as inadequate measures in court, and the public in general will not accept ineffectual management practices. TMDLs can provide the mechanism to effectively reduce nonpoint source pollution in waterways.

While risk assessment is an important tool in environmental analysis regarding health risks, its application to TMDLs is limited by the fact that the state has already established acceptable levels of coliform and enterococcus for coastal waters. That is not to say that health standards are established beyond a reasonable doubt and that there is no further need for risk analysis. However, the key regulatory issues at this time for TMDLs for coastal bacteria contaminants is the identification of pollutant sources and establishing load allocations for these sources.

With regard to other types of pollutants that migrate to coastal waterways, TMDLs will have a wide potential for application for pollutants such as nitrogen, phosphorus

and heavy metals. Federal, state and local agencies developing TMDLs for these pollutants will certainly examine risk assessment studies as one factor in developing acceptable levels of pollutants. While the Clean Water Act has established ambient levels for point sources, ambient levels for nonpoint sources will be established in effect under TMDLs.

5.7   Development of TMDLs

It should be apparent that the development of a TMDL for Arroyo Burro will be a long and complex process. However, the task is not insurmountable. The basic blue print for the development of a TMDL is as follows:

1. **Problem Statement.**   This includes a description of the water body or watershed setting, and an identification of the beneficial use impairments of concern and the pollutants or stressors causing these impairments.

2. **Numeric Target(s).** Limits must be set for indicators of each pollutant or stressor addressed in the TMDL.

3. **Source Analysis.**   This includes an assessment of the relative contributions of pollutant sources or impairment stressors, and the extent to which these sources and stressors must be controlled.

4. **Loading Capacity Estimate.** This is an estimate of the assimilative capacity of the water body for the pollutants(s) of concern.

5. **Allocations. A**llocations of allowable loads or load reductions must be set among different sources of concern, providing an adequate margin of safety. These allocations are expressed as wasteload allocations to point sources and load allocations to nonpoint sources. The TMDL equals the sum of these allocations and cannot exceed the loading capacity of the water body.

6. **Monitoring Plan.**   This plan would monitor the effectiveness of TMDLs and if necessary, would include a review and revision of TMDLs (Ruffolo April 1999 p.12).

The initial step to be taken by the county is the development of a water quality standard for the impaired waterway of Arroyo Burro. This includes the designation of the specific beneficial use of the waterway, such as swimming and bathing. An ocean standard exists under the Ocean Plan for maximum levels of total and fecal coliform in near shore ocean waters (California State Code 1994). The Water Board's Basin Plan sets fecal coliform levels for watersheds at 200/100 ml, (Regional Water Quality Control Board 1994) which is the same level as in the Ocean Plan. Therefore, it appears that the standard will be consistent for both ocean and creek waters. According to the State TMDL Coordinator, Stefan Lordezato, a water standard may

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consist of the beach standard, the drainage basin standard, or a combination of both (Lordezato 1999). This issue will need to be addressed by the county and/or Regional Water Quality Control Board. It should be noted that the EPA has delegated authority for the development of a TMDL to the Regional Water Quality Control Board, which in turn has the discretion to delegate to the county (Lordezato 1999). Whether the water board will delegate this responsibility to the county remains to be seen. However, even if the authority to develop a TMDL is delegated, the TMDL still must be approved by the Regional Water Quality Control Board.

The next step is to identify sources of pollution in the waterway that are contributing to bacterial contamination in Arroyo Burro. The county has begun efforts to identify these sources, but the possibilities are numerous. Storm drains, septic tanks, animal and human feces runoff, and other possible sources complicate this process. The county will find this a challenging process, but it is critical to the development of a TMDL.

Once the sources are identified, then numeric targets are established in order to evaluate whether the water standard is being met. One option available is to employ a more concise tracking method using DNA methods. A TMDL must allocate load limits of bacteria between the various sources, and this requires a determination of how much bacteria can exist and which sources should be allocated load limits (Lordezato 1999).

The overall allocation of the total amount of bacteria permitted in the waterway may consist of a "not to exceed" figure within a certain time period. The allocation to specific sources, such as storm drain discharges and septic tanks, would take into account variability and frequency, since bacterial contamination is variable in time and space (Lordezato 1999).

Another option is to write the TMDL in such a way that narrative standards are utilized. For example, the standard may be "no adverse impact on human health created by discharge." This is a much more general standard than numeric limits and its effectiveness on improving water quality is uncertain.

In at least one case in California, the development of a bacterial TMDL has proven to be such a challenge that Newport Bay has commissioned further studies before any load allocations are developed. Given the complexity of sources to Arroyo Burro, the county and/or water board may very well select this alternative. Animal/human feces will present the greatest challenge of sampling and identification because the sources are not stationary.

The key throughout the process of developing a TMDL is insuring that the allocations made will result in the attainment of the water standard (Lordezato 1999). This should be the guiding principle in the preparation of a TMDL.

It is clear from the foregoing discussion that the TMDL process is complex and will require an enormous amount of effort. The establishment of the water standard for Arroyo Burro will be relatively straight forward compared to the identification of sources and allocation of bacterial load limits. However, as the process is refined, more knowledge will become available as TMDLs are developed throughout the country. One thing for certain is that "it will be many years before California sees TMDL limits implemented, especially for nonpoint sources, unless the courts intervene with new capacity (Ruffolo 1999)." Needless to say, it is quite likely that "water quality regulation is in the act of changing in fundamental ways" (Ruffolo 1999) as a result of the sweeping powers of TMDLs.

6.0   Development of Short and Long Term Best Management Practices

As mentioned previously, Santa Barbara County is currently undergoing the process of applying for an NPDES permit under the Clean Water Act. As part of the NPDES permit process, the County is required to implement best management practices (BMPs) to reduce pollutant loading for municipal sites, new construction sites, and new developments and redevelopment sites. In the absence of a comprehensive TMDL program, the County's adherence to the NPDES Phase II requirements is an initial action policy option to address the pollution problem. In this section of the report we discuss specific best management practices that aim to reduce bacterial loading for both watersheds, and how the findings from our various scientific analyses relate to the selection of BMPs.

It is unclear how well equipped the NPDES BMP requirements are to deal with diffuse nonpoint sources of bacterial pollution in watersheds. In order to attempt to alleviate the growing problem of nonpoint source pollution, Congress passed section 6217 of the Coastal Zone Act Reauthorization Amendments (CZARA) in 1990. These amendments require coastal states to adopt EPA specified guidelines that address the reduction of nonpoint pollution in coastal waters. These published guidelines specify "management measures to restore and protect coastal waters from specific types of nonpoint source pollution." Management measures are legally defined as "economically achievable measures that reflect the best available technology for reducing pollutants" (USEPA 1993). The EPA specified measures are separated into a number of different categories such as: new development management measures, existing development measures, pollution prevention measures, watershed restoration measures, measures for roads, highways, and bridges, etc. As stated previously in the legal analysis section, the enforcement capabilities of section 6217 CZARA remain questionable. However, if a local authority chooses to meet these requirements in addition to or as part of CWA best management practice requirements, we believe that this would provide the best mechanism to reduce bacterial loading in coastal watersheds. Furthermore, development of TMDL allocations may drive the implementation of best management practices and provide a good tool for integrating point and nonpoint source pollution controls.

Management measures under CZARA are broad categories under which specific "management practices" are to be adopted for the reduction of nonpoint source pollution to coastal areas. There are two types of management practices: preventive measures (source controls) and delivery reduction measures. Preventive measures are aimed at reducing the exposure of pollutants to storm water. Generally, they are long-term solutions to water quality problems. Preventative measures could include implementing ongoing public awareness campaigns, developing onsite disposal system ordinances, maintenance and monitoring of sewer line integrity, and undertaking watershed scale management programs. Delivery reduction measures are specific management practices aimed at reducing the amount of pollution that has already been incorporated into stormwater from reaching beaches. They are generally short-term solutions. Some examples of delivery reduction measures include installing pollution interception devices such as sand filters, infiltration trenches, sediment ponds, and vegetative buffer zones along creek margins (EPA 1993). It is usually necessary to implement a series of preventive and delivery reduction measures to achieve significant pollution reductions.

To clarify the relationship between the CZARA and the NPDES requirements, the EPA guidelines state that any stormwater regulated by a NPDES Phase II permit will no longer be subject to the CZARA requirements. Nevertheless, we feel that the management practices recommended by the EPA guidance report are highly relevant and we urge the County to consider them in their attempt to formulate a comprehensive management plan.

The County report to the Board of Supervisors on Project Clean Water describes several recommended future actions to address the problem of bacterial contamination (URS Greiner Woodward-Clyde 1999). The report describes many short-term and long-term potential management measures. We have outlined a few short-term and long-term solutions that we feel are consistent with the results of our analyses. Our recommendations are structured to augment the findings and recommendations of the County.

6.1   Short-Term Management Practices

In its search for potential solutions to the marine water contamination problem, the County hired the consultant firm of URS Greiner Woodward-Clyde to undertake a feasibility study of short-term temporary treatment alternatives for contaminated stormwater in the Arroyo Burro watershed in 1998-1999. In this report, Woodward-Clyde examines a series of potential short-term, "end-of-pipe" treatment alternatives aimed at reducing the amount of bacteria in stormwater runoff that reaches the near shore lagoon at Arroyo Beach. In the report, Woodward-Clyde states that the County indicated that any "end-of-the-pipe" treatment alternative considered "would be applied as a short-term (i.e., seasonal only) and temporary (i.e. several years only) solution until the long-term source reduction measures are effective" (URS Greiner

Woodward-Clyde 1999). The treatment alternatives examined in this report were intended for use only during the summer months during low creek flow regimes.

The report suggests that the most promising stormwater treatment alternatives for the Arroyo Burro watershed involve in situ stormwater treatment via UV and ozone disinfection devices (URS Greiner Woodward-Clyde, 1999). Based on specific space requirements and site conditions, Woodward-Clyde found that the most feasible location of the stormwater treatment facility would be upstream from the lagoon at Hendry's Beach. The treatment alternatives examined would treat surface stormwater before it reaches the downstream lagoon. Furthermore, the treatment alternatives examined allow sediments to pass through the treatment facility to the creek reach below.

Our analysis shows that freshwater samples containing sediment may contain higher levels of coliform bacteria than samples with lesser amounts of sediment. Ozonation provides a greater potential to reduce the amount sediment-bound bacteria found in the water column than UV disinfection (Bitton 1994). If following treatment, sediment particles still contain significant amounts of bacteria, it may be necessary to trap these particles before they reach the lagoon and are eventually discharged to the ocean. The EPA CZARA guidelines describe several devices that reduce the amount of sediments in stormwater to varying degrees (EPA 1993). Two of the more effective structural devices that remove sediment particles from stormwater are sand filters and infiltration trenches/basins. Appendix XV shows diagrams of these types of filtration devices. Installation of these structural measures would be subject to the same permit requirements mentioned in the Woodward-Clyde report (1999).

Infiltration trenches are 3-8 foot deep trenches that are filled with stones that are aimed at increasing the amount of groundwater recharge. As stormwater flows into these trenches, it filters down to the bottom of the trench, straining sediments through the stone media, ultimately allowing increased water infiltration into the ground. Infiltration devices, when operating correctly, have bacteria removal rates between 75%-98%, and 95% removal rates for sediment (Schueler *et al.* 1992). Our hydrological analysis showed that increasing the amount of pervious land use results in reduced runoff and sediment loading. Infiltration trenches promote groundwater infiltration and can be employed to counteract the deleterious effects of impervious land use. However, infiltration devices have been known to have high failure rates. Slightly over half of all infiltration trenches fail within 5 years of installation (Scheuler *et al.* 1992). Moreover, if soils beneath the trench are not porous enough, or the water table is very high, infiltration trenches can fill up and lose their purpose. Infiltration devices must be inspected regularly and mowed periodically to remove woody plant matter from the surface and cleared of sediment that choke the trench. Furthermore, infiltration trenches should be used in conjunction with other sediment settling management practices to reduce the probability of clogging. Schueler estimates annual maintenance costs at being 5 to 10 percent of capital costs (Schueler

et al. 1992). Also, if an infiltration device becomes clogged, it may have to be completely rebuilt.

A variation of the infiltration trench is the infiltration basin. Like infiltration trenches, infiltration basins aim at increasing the amount of groundwater recharge while sifting suspended solids out of stormwater. A key feature of the infiltration basin is an exfiltration storage area where sediments can settle before stormwater exits the device. This bay region is usually covered with grass turf. However, like infiltration trenches, infiltration basins have relatively high failure rates within the first 10 years of operation and require regular maintenance (Scheuler *et al.* 1992). If used as a short-term solution, the expected operational life of such devices may be sufficient.

Sand filters work by filtering stormwater through beds of fine sand and may be installed underground or in self-contained concrete boxes. As stormwater filters through the sand bed, sediment and bacteria adsorb onto the sand particles, and the effluent stormwater leaves through pipes located on the bottom of the device. Sand filters have shown the following monitoring performance removal rates: 85% for sediment, 35% for nitrogen, 40% for dissolved phosphorus, and 40 % for fecal coliform (Scheuler *et al.* 1992). Sand filters can become clogged relatively frequently, yet the maintenance is fairly simple. Maintenance involves raking the sand and removing any other matter that could be clogging the filtration ability of the device. Sand filters are more expensive to construct than infiltration trenches and are less effective at removing coliform bacteria (NCSU Water Quality Group, 1999).

Unfortunately, infiltration devices may remove flora and fauna that are not a threat to human health and are critical to the lagoon and marine ecosystems. It is for this reason that we suggest these types of devices to be used only as a short-term solution. Long-term use of any treatment device could adversely impact natural systems by altering floral and faunal species composition.

These types of filtration devices do not necessarily need to be used in conjunction with a stormwater disinfection device. These types of structural filtering devices can be used by themselves or in series. The implementation of these devices could serve as a lower cost alternative than disinfection devices (Schueler *et al.* 1992), however their effectiveness varies. Moreover, infiltration devices must be cleaned regularly. Implementing structural measures such as sand filters and infiltration trenches/basins would require a strong commitment by county officials to establish an effective maintenance schedule.

These stormwater best management measures, like those evaluated by URS Greiner Woodward-Clyde, would treat stormwater before it reaches the lagoon environment. These structural measures would not prevent any fecal deposition into the lagoon ecosystem from waterfowl utilizing lagoon waters. This factor could impose serious limitations on the effectiveness of these BMPs. Additionally, any impacts caused by these structural measures on flora and fauna living in the brackish lagoon waters

would need to be assessed before various permit are granted. A comprehensive list of relevant permit requirements can be found in the URS Greiner Woodward-Clyde report (1999).

6.2   Long Term Solutions

As part of their report to the Santa Barbara County Board of Supervisors, the Project Clean Water stakeholders recommended against adopting the short-term solutions analyzed by Woodward-Clyde for the time being (Santa Barbara County Water Resources 1999). This was due to the fact that the solutions would be effective only under low flow conditions and the permitting process would delay the implementation of these solutions until after the summer of 1999. The Project Clean Water report did recommend long-term solutions for creek and beach bacterial pollution. Generally these are source reduction measures aimed at reducing the amount of pollutants that reach the creeks and beaches. The following is a summarized list of potential long-term strategies aimed combating coastal bacterial contamination that were proposed in the county report (Ibid).

- A review and possible revision of existing ordinances, regulations, and codes designed to improve creek and beach water quality.
- An investigation of possible sites for wetland and riparian restoration in order to decrease pollutant loading to creeks.
- Storm drain, creek, and street cleaning, and possible retrofitting of storm drains with filtration devices.
- Strategies such as education regarding the prevention of domestic and grazing animal feces from contaminating watersheds and beaches.
- Increased inspections for illicit connections and discharges to storm drains.
- Mapping and testing of septic systems to ascertain possible sources of groundwater pollution, and upgrading failing septic systems.
- Public outreach including community education and involvement.
- Control and monitoring of municipal operations, development projects, and commercial and industrial facilities, perhaps including the implementation of specific BMPs such as ordinances requiring retention ponds and erosion control.

In addition, the county and city are interested in investigating the effects of lateral migration of bacterial contaminants away from the Arroyo Burro creek mouth during low flow conditions to better determine the extent of beach closures. Finally, the county, in cooperation with Heal the Ocean, is planning to conduct a study using DNA analysis to better determine sources of bacterial contamination. This study is to be conducted at Rincon beach with the hope that the results can be applied to other Santa Barbara beaches.

Developing long term solutions to bacterial pollution is an enormous task, as it requires a thorough understanding of watershed processes and their affect on the

transport of bacterial pollution. In order to address the complex issues associated with abating bacterial pollution, we recommend further study in three areas:

> 1. Research in the area of DNA detection methods in order to help pinpoint sources of bacterial pollution and gain more insight into the actual pathogen assemblages in urban stormwater.

> 2. Research to determine if lagoon sediments actually harbor high concentrations of bacteria in order to determine if sediment accumulation in lagoon sediments could be a major source of bacteria to near shore waters upon release of these sediments to the marine environment.

> 3. Further research in the area of hydrology on the effects of land use and imperviousness on bacterial loading. This research could help incorporate watershed modeling analyses into land use planning decisions, TMDL development, and effective BMP implementation.

Further Research on DNA Sampling Techniques

From our review of the relevant literature and the results of our analyses, it seems clear that the current methods of pathogen detection are inadequate for several reasons. First, current coliform testing methods do not detect the origin of the waste in stormwater. This is especially problematic because these testing methods do not allow officials to precisely pinpoint the source of the bacterial loads. The DNA fingerprinting methods described earlier in our report could be a powerful method to pinpoint load sources and eventually aid in the development of load allocations.

Furthermore, DNA testing methods could also help determine the actual pathogen assemblage in marine waters, and thus facilitate a more accurate analysis of the risk of swimming at contaminated beaches. From our risk analysis, it was apparent that current testing methods may understate or overstate the pathogenic potential of marine water samples. The pathogenic potential of water samples depends on whether positive tests result from organisms originating from human or animal waste, or if they result from non-pathogenic indigenous organisms, thus giving false positive sampling results. DNA fingerprinting could provide better insight into how pathogenic Santa Barbara's coastal waters actually are. However, much of this research is new and will require time and effort to perfect. It seems that Santa Barbara County would be an excellent place to continue research of this kind due to the apparent commitment of the public, local agencies, and research interests to finding a solution to the problem.

Further Study of Pathogen Content of Lagoon Sediments

Through our sampling analysis and our review of the literature, freshwater samples with sediments may contain higher concentrations of coliform bacteria. However, it

remains unclear whether or not lagoon sediments are actually a source or a sink of coliform bacteria. Lagoon sediments may contain higher or lower concentrations of bacteria depending on the fate of sediment bound bacteria in the lagoon environment. Consequently, we recommend that future studies focus on determining the pathogenic potential of lagoon sediments, and whether sediments leaving the lagoon environment to the marine environment are a significant source of coliform bacteria. Such studies may choose to incorporate DNA analyses as well, in order to more precisely determine the types of organisms contained in the lagoon environment and their pathogenic potential.

Further Research on Land-Use Effects on Bacterial Loading

Studies have shown that land use composition can have a dramatic effect on pollutant transport throughout watersheds. Specifically, high amounts of impervious land in urban watersheds can increase the likelihood of increased pollutant transport (Ventura and Kim 1993). It is imperative to understand the relationship between land use distributions and pollutant loading in urban watersheds in order to make well-informed land use decisions to minimize pollution impacts. The use of computer software packages that couple digital geographic database information with pollutant loading simulators can be a very powerful tool for urban planners.

One such system that has been created by the EPA is known as Better Assessment Science Integrating Point and Nonpoint Sources (BASINS). The BASINS system integrates geographic information system (GIS) data from a number of different database sources with three different water quality models: Nonpoint Source Model (NPSM), Qual2E, and Toxiroute (EPA 1998c). National GIS database information includes information such as: cartographic/topographic information (USGS), soil type characteristics (USGS), precipitation data (NOAA), point source pollution data for a number of pollutants (USEPA), and water quality monitoring station data (USEPA). The stream water quality models simulate pollutant loading for many pollutants throughout delineated watersheds by integrating the various data layers into the calculations.

NPSM is an appropriate model for studying how pollutant loads vary with different land use composition scenarios. NPSM has the capability to model surface and groundwater pollutant loading based on the following specific watershed land use categories: urban or built-up land, pasture land, agricultural land, range land, forested land, and barren land. Each land use can be assigned a specific "percent perviousness." This percentage describes what area of the particular land use is pervious to stormwater infiltration. In addition, stream reach characteristics such as slope, mean/max depth, Manning's roughness, and flood plain width incorporate specific creek characteristics into the model. Ultimately, by combining the various GIS database information with the creek characteristic inputs, NPSM models pollutant

loading levels for any historic period for which precipitation data is available (typically 1970-1995). NPSM allows the user to reclassify land uses to effectively increase or decrease the "percent perviousness" of the watershed landscape. NPSM can then calculate different pollutant loading scenarios based on a variety of simulated land use changes. Hence, the BASINS system is an analytical tool that can allow land use planners to better understand the relationships between future land use change and pollution impacts, and possibly use this information in for implementing future land use policies.

As a long term planning strategy, the county might wish to consider amending the Santa Barbara zoning ordinances in order to reduce the amount of urban land use in watersheds that still have room for such growth. The zoning ordinances provide for the classification and regulation of the uses of land, buildings and structures within Santa Barbara County. The zoning ordinances also designate various categories of districts to include agricultural, residential, commercial, and industrial (Santa Barbara County 1996).

The zoning ordinances can be amended by a process that includes public hearing, review by the planning commission, and approval of the amendment by the board of supervisors (Santa Barbara County Planning Division 1990). If it could be conclusively shown that non-porous surfaces in urban area are contributing to coastal bacterial contamination, there is a possibility of amending the zoning ordinances to take into account this source of pollution. Amending the zoning ordinances would have to take into account the various residential and commercial zoning districts that would each need to be modified. In addition, there may be an issue of "land taking" if future development is prohibited by the amendment. Furthermore, there would likely be political opposition to such an amendment by business interests. This strategy would be an involved and lengthy process that would involve many stakeholders. It is unclear as to how effectively scientific knowledge could influence land use planning decisions that are generally wrought by political influence.

In addition to providing insight into how land use affects pollutant loading, systems like BASINS can help environmental planners establish TMDLs and determine optimal locations for BMP placement. Local pollution data (from point sources and specific sampling points) can be visualized spatially in GIS packages. Models such as NPSM that use data such as stream characteristics and groundwater transport are powerful tools that help in understanding the transport of bacteria throughout watersheds. Improved modeling would help determine better sampling regimes for the development of TMDLs and would help determine proper zoning ordinances and land use restrictions. These simulations can be used to aid the development of water quality monitoring schemes, and to determine key hot spot pollution areas. Pollution data updates help calibrate water quality models and help establish more accurate pollution loading simulations. As mentioned before, ongoing watershed monitoring programs must be developed in order to effectively enforce TMDLs. Moreover, an analysis of spatially distributed data can help determine the optimal configuration of BMPs based

on where pollutant loads are highest, and what types of pollutants are problematic in certain watershed areas.

7.0 Conclusions

We have developed several conclusions as a result of our research of the bacterial pollution problem in Santa Barbara County. Each section of this report developed conclusions that were stated earlier in the various sections. The summary of these conclusions follows.

Through our legal analysis, we argued that the most effective manner in which to reduce bacterial pollution was through the development of Total Maximum Daily Loads (TMDLs). TMDLs define specific pollution load allocations that may be released from point or nonpoint sources within watersheds. TMDLs are a more effective management tool than Best Management Practices (BMPs) because TMDLs are enforceable, while BMPs are not. Therefore, we recommend that TMDLs should eventually be determined for Santa Barbara County watersheds. However, the development of TMDLs is not a trivial process. It is extremely data intensive and requires that the scientific data be valid and reliable. Given the problems with accurately measuring pathogens in waters, current testing techniques may not be reliable enough to enforce compliance.

Our risk analysis predicted the seasonal risks of swimming at Arroyo Burro beach according to the amount of fecal streptococcus measured. The risk of swimming at Arroyo Quemada does not depend on the season. Our calculations show that potential risks of contracting gastroenteritis were higher for the wet seasonal periods than for the dry seasonal periods at Arroyo Burro Beach for each dose-response relationship used. In comparison, illness rate predictions at Arroyo Quemada were higher than Arroyo Burro for all seasons. The predicted illness rates at Arroyo Quemada were higher than the highest illness rates calculated for the wet 1997-1998 seasonal period at Arroyo Burro. In addition, we found differences between the calculated illness rates from one dose-response relationship to the next. Furthermore, when we accounted for uncertainty in the calculations using the two randomized controlled study relationships, the standard deviations for the calculations were quite large.

Through our microbiology research, we determined that water samples containing creek sediments contain higher levels of total coliform than water samples without sediment. We also determined that bacteria are strongly positively correlated with precipitation during the rainy season of Santa Barbara County. Our fertilizer application analysis showed that in 1998 there were higher levels of total coliform present in water samples after the application of fertilizer.

The SCS rainfall-runoff model predicted that a one-inch precipitation event during dry soil conditions will produce little runoff, but the same event during wet conditions

will produce more runoff, and thus sediment loading, which can increase bacterial loading into coastal waters. Arroyo Burro produced more runoff than Arroyo Quemado for the same precipitation event, even when the different sizes of the two watersheds were taken into account. This was due to the greater degree of urbanization and higher curve numbers associated with developed land uses in the Arroyo Burro watershed.

We examined both short term and long term best management practices aimed at reducing sediment and bacterial loading to watersheds and beaches. We presented county solutions as well as our own recommendations, based on our previous research. We recommend further research in the areas of DNA sampling techniques, pathogen content of lagoon sediments, and land use effects on bacterial loading, in order to better determine effective BMPs.

We used an integrated study of the coastal bacterial contamination problem of Santa Barbara County to examine potential solutions to this complex issue. The scientific research results helped in the analysis of various BMPs and the legal analysis determined which enforcement methods of reducing the contamination might be the most useful. Throughout the report, we discuss future areas of research that will aid in the determination of feasible solutions to the bacterial contamination problem. Among these, two important areas are the development of more accurate cost-effective measurements of pathogens in water, and the development of pathogen source detection techniques. These two key areas will aid in calculating more accurate estimations of swimming-related health risks, and will provide tools for effective enforcement of nonpoint sources of pollution through the development of TMDL programs.

The development of a successful pollution control program must bear in mind all relevant scientific data, economic, and socio-political issues. As is the case with most environmental problems, the problem of coastal bacterial pollution is one that encompasses many scientific, legal, and political issues. Given the complexity of nonpoint source pollution, we believe that an integrative approach such as ours is the most useful means in which to study this multi-faceted problem.

8.0 Acknowledgments

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Appendix I: Project Site Map

Arroyo Quemada
Beach

UCSB

Arroyo Burro
Beach

Legend:
★ Beaches
★ UCSB
Streams/Creeks
landuse
Urban or Built-up Land
Agricultural Land
Rangeland
Forest Land
Water
Wetland
Barren Land
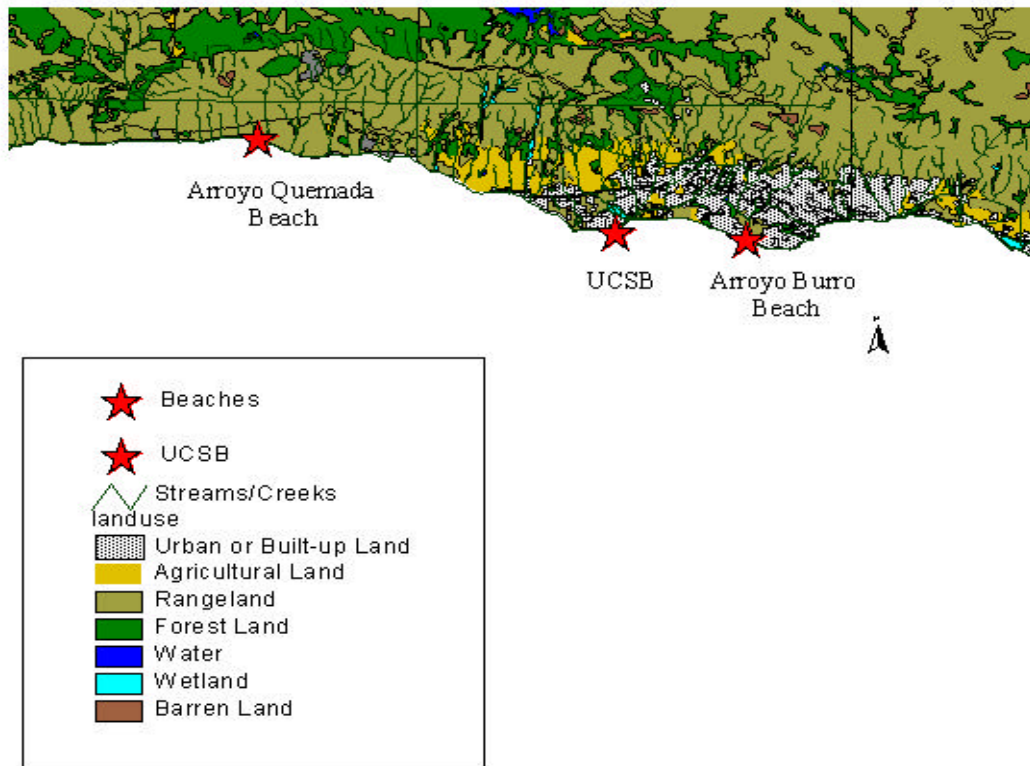
Source: BASINS (EPA 1998)

Appendix II: Graphical illustration of the relationship between precipitation and bacterial counts

Figure 1: Cumulative Weekly Precipitation and Total Coliform Counts at Arroyo Burro Beach from Oct. 1996 to Apr. 1998

Figure 2: Precipitation and Total Coliform Counts at Arroyo Quemado Beach from Nov. 1997- Apr. 1998
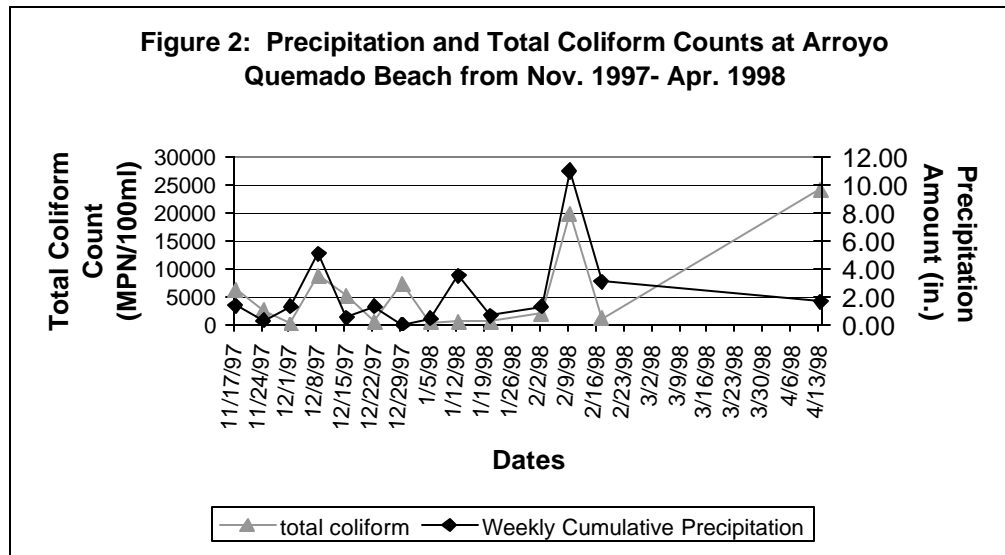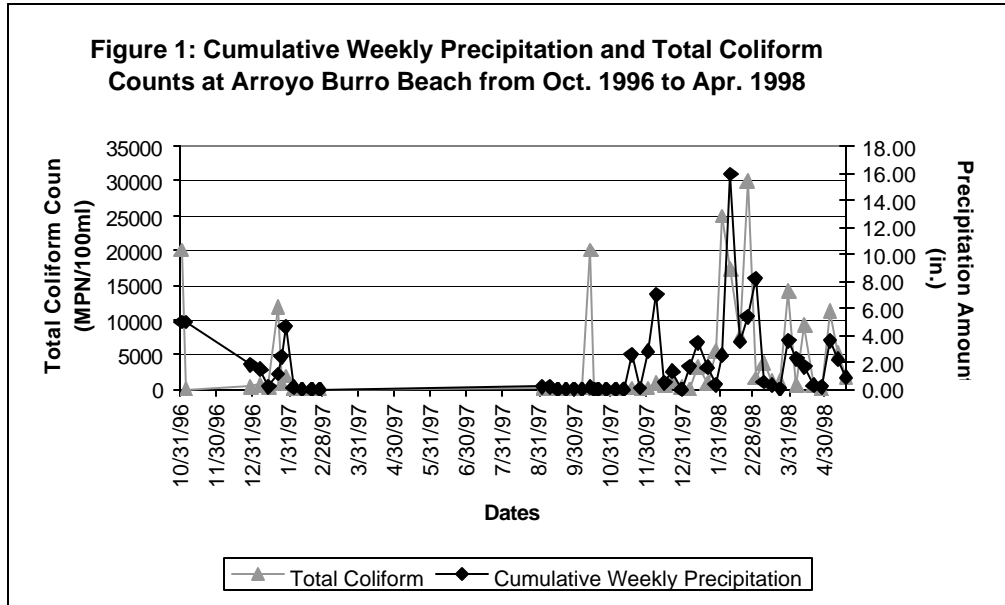
Appendix III: Rank tables for Arroyo Quemada and Arroyo Burro

Table 1: Ranked Total Coliform and Precipitation for Arroyo Quemada

date	cum. week precipitation	total coliform	rank precip	Rank tc	D	D^2	sum D^2	r_s
11/17/97	1.32	6240	8	10	-2	4	260	.43
11/24/97	0.25	2540	9	8	1	1		
12/1/97	1.25	200	5	1	4	16		
12/8/97	5.05	8664	13	12	1	1		
12/15/97	0.48	5172	3	9	-6	36		
12/22/97	1.30	482	7	4	3	9		
12/29/97	0.00	7270	1	11	-10	100		
1/5/98	0.40	295	2	2	0	0		
1/12/98	3.48	565	12	5	7	49		
1/20/98	0.62	399	4	3	1	1		
2/2/98	1.27	1956	6	7	-1	1		
2/9/98	10.98	19863	14	13	1	1		
2/17/98	3.05	985	11	6	5	25		
4/13/98	1.63	24192	10	14	-4	16		

Table 2: Ranked Total Coliform and Precipitation for Arroyo Burro

Date	Cum week precip	Total Coliform	rank precip	Rank tc	Rt pre-cip	Rt tc	D	D^2	Sum D^2	Rs
10/31/96	4.99	20050	30	31	38.8	39.2	-0.4	0.16	5592	.75
11/4/96	4.99	10	30	1	38.8	6.2	32.6	1063		
12/29/96	1.81	359	20	6	27.3	14.2	13.1	171.6		
1/6/97	1.46	640	17	11	23.8	19.2	4.6	21.16		
1/13/97	0.20	420	8	7	14.8	15.2	-0.4	0.16		
1/21/97	1.13	11840	15	28	21.8	36.2	-14.4	207.4		
1/24/97	2.45	750	23	12	30.3	20.2	10.1	102		
1/27/97	4.64	1920	29	20	37.3	28.2	9.1	82.81		
2/3/97	0.17	10	7	1	12.3	6.2	6.1	37.21		
2/10/97	0.00	100	1	3	6.3	9.2	-2.9	8.41		
2/18/97	0.00	100	1	3	6.3	9.2	-2.9	8.41		
2/25/97	0.01	10	2	1	7.3	6.2	1.1	1.21		
9/2/97	0.20	100	8	3	14.8	9.2	5.6	31.36		
9/8/97	0.20	10	8	1	14.8	6.2	8.6	73.96		
9/15/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
9/22/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
9/29/97	0.04	10	4	1	9.3	6.2	3.1	9.61		
10/6/97	0.00	200	1	5	6.3	12.7	-6.4	40.96		
10/13/97	0.14	20100	6	32	11.3	40.2	-28.9	835.2		
10/16/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
10/20/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
10/27/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
11/3/97	0.00	10	1	1	6.3	6.2	0.1	0.01		
11/10/97	0.00	200	1	5	6.3	12.7	-6.4	40.96		
11/17/97	2.58	200	25	5	32.3	12.7	19.6	384.2		
11/24/97	0.10	10	5	1	10.3	6.2	4.1	16.81		
12/1/97	2.82	200	26	5	33.3	12.7	20.6	424.4		
12/8/97	7.01	983	32	14	40.8	22.2	18.6	346		
12/15/97	0.48	627	12	10	18.8	18.2	0.6	0.36		
12/22/97	1.33	1106	16	15	22.8	23.2	-0.4	0.16		
12/29/97	0.00	359	1	6	6.3	14.2	-7.9	62.41		
1/5/98	1.66	148	18	4	25.3	10.2	15.1	228		
1/12/98	3.43	3255	27	21	34.8	29.2	5.6	31.36		
1/20/98	1.64	886	18	13	25.3	21.2	4.1	16.81		
1/27/98	0.35	5475	11	24	17.8	32.2	-14.4	207.4		
2/2/98	2.53	25000	24	33	31.3	41.2	-9.9	98.01		

Table 2 continued: Ranked Total Coliform and Precipitation for Arroyo Burro

Date	Cum week precip	Total Coli-form	rank pre-cip	Rank tc	Rt precip	Rt tc	D	D^2	Sum D^2	Rs
2/9/98	15.95	17329	34	30	42.8	38.2	4.6	21.16	5592	.75
2/17/98	3.53	7701	27	25	34.8	33.2	1.6	2.56		
2/23/98	5.39	30000	31	34	39.8	42.2	-2.4	5.76		
3/2/98	8.20	1616	33	18	41.8	26.2	15.6	243.4		
3/9/98	0.55	3873	13	22	19.8	30.2	-10.4	108.2		
3/16/98	0.28	1160	9	17	15.8	25.2	-9.4	88.36		
3/23/98	0.03	1145	3	16	8.3	24.2	-15.9	252.8		
3/30/98	3.61	14136	28	29	36.3	37.2	-0.9	0.81		
4/6/98	2.27	620	22	9	29.3	17.2	12.1	146.4		
4/13/98	1.68	9208	19	26	26.3	34.2	-7.9	62.41		
4/20/98	0.32	496	10	8	16.8	16.2	0.6	0.36		
4/27/98	0.20	98	8	2	14.8	7.2	7.6	57.76		
5/4/98	3.57	11199	28	27	36.3	35.2	1.1	1.21		
5/11/98	2.20	5172	21	23	28.3	31.2	-2.9	8.41		
5/18/98	0.90	1793	14	19	20.8	27.2	-6.4	40.96		

Appendix IV: Graphical summary of sampling results

Figure 1: Sampling Results - Total Coliform Counts at Pila Creek

Figure 2: Sampling Result - *E.coli* Counts at Pila Creek
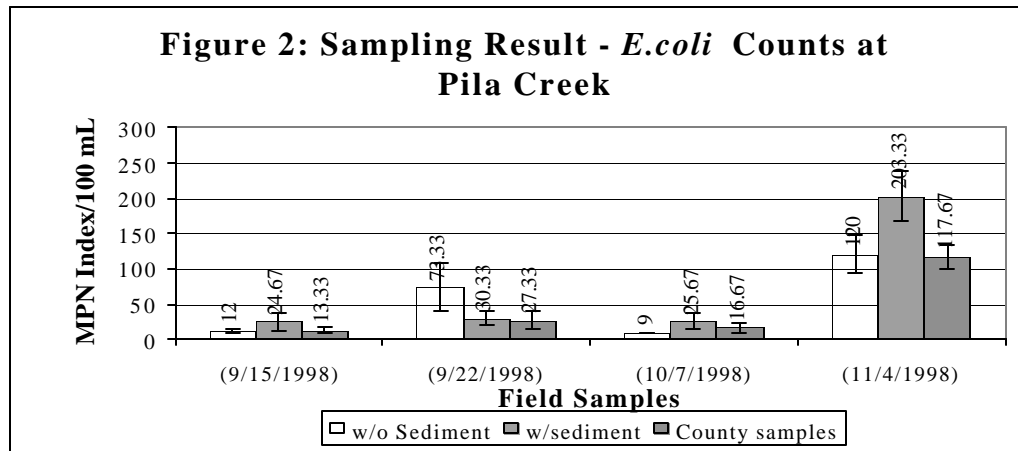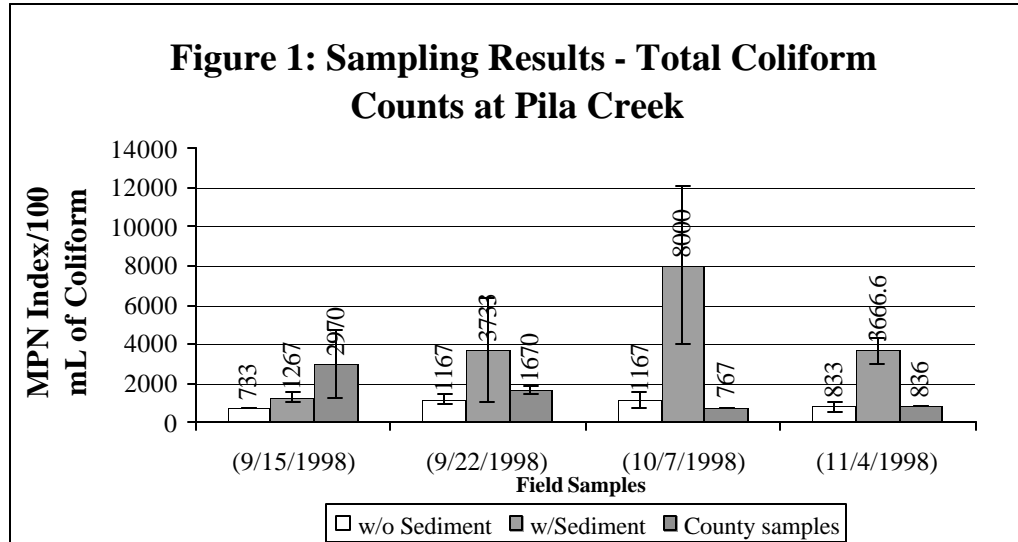
Appendix V: Statistical summary table of sampling results

Date	9/15/98			*9/22/98			10/7/98			11/5/98		
	w/o Sediment	w/ Sediment	County samples	w/o Sediment	w/ Sediment	w/o Sediment	w/o Sediment	w/ Sediment	County Samples	w/o Sediment	w/ Sediment	County Samples
	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform	Total Coliform
Replicates												
Rep. 1 (MPN)	700	1700	6488	900	1700	1421	1600	3000	766	700	5000	833
Rep. 2 (MPN)	800	1300	906	900	500	2014	1600	>16000	723	1300	3000	836
Rep. 3 (MPN)	700	800	1515	1700	9000	1576	300	5000	813	500	3000	839
Mean (of replicates)	733	1270	2970	1170	3730	1670	1170	8000	767	833	3670	836
Standard Error	33.3	260	1770	267	2660	178	433	4040	26.0	240	667	1.73
Median	700	1300	1520	900	1700	1580	1600	5000	766	700	3000	836
Standard Deviation	57.7	451	3060	462	4600	308	751	7000.00	45.0	416	1150	3.00
Sample Variance	3330	203000	9380000	213000	21200000	94600	563000	49000000	2030	173000	1330000	9.00
Confidence Level(95.0%)	143	1120	7610	1150	11400	764	1860	17400	112	1030	2870	7.45

Appendix V continued

Date	9/15/98			*9/22/98			10/7/98			11/5/98		
	w/o Sediment	w/ Sediment	County samples	w/o Sediment	w/ Sediment	w/o Sediment	w/o Sediment	w/ Sediment	County Samples	w/o Sediment	w/ Sediment	County Samples
	E. coli	E.coli	E. coli	E. coli	E.coli	E. coli	E. coli	E.coli	E. coli	E. coli	E.coli	E. coli
Replicates												
Rep. 1 (MPN/100mL)	17.00	50	20	50	50	10	8	13	30	110	130	145
Rep. 2 (MPN/100mL)	8.00	11	10	30	11	52	8	14	10	170	240	122
Rep. 3 (MPN/100mL)	11.00	13	10	140	30	20	11	50	<10	80	240	86
Mean (of replicates)	12.0	24.7	13.3	73.3	30.3	27.3	9.00	25.7	16.7	120	203	118
Standard Error	2.65	12.7	3.33	33.8	11.3	12.7	1.00	12.2	6.67	26.5	36.7	17.2
Median	11.00	13	10	50	30	20	8	14	10	110	240	122
Standard Deviation	4.58	22.0	5.77	58.6	19.5	21.9	1.73	21.1	11.6	45.8	63.5	29.7
Sample Variance	21.0	482.3	33.3	3433.3	380.3	481.3	3.00	444	133	2100	4030	884
Confidence Level(95.0%)	11.4	54.6	14.3	146	48.4	54.5	4.30	52.4	28.7	114	158	73.9

*9/22/98 sampling method differs from the others because the top 10 cm of sediment was stirred instead of the top 2 cm.
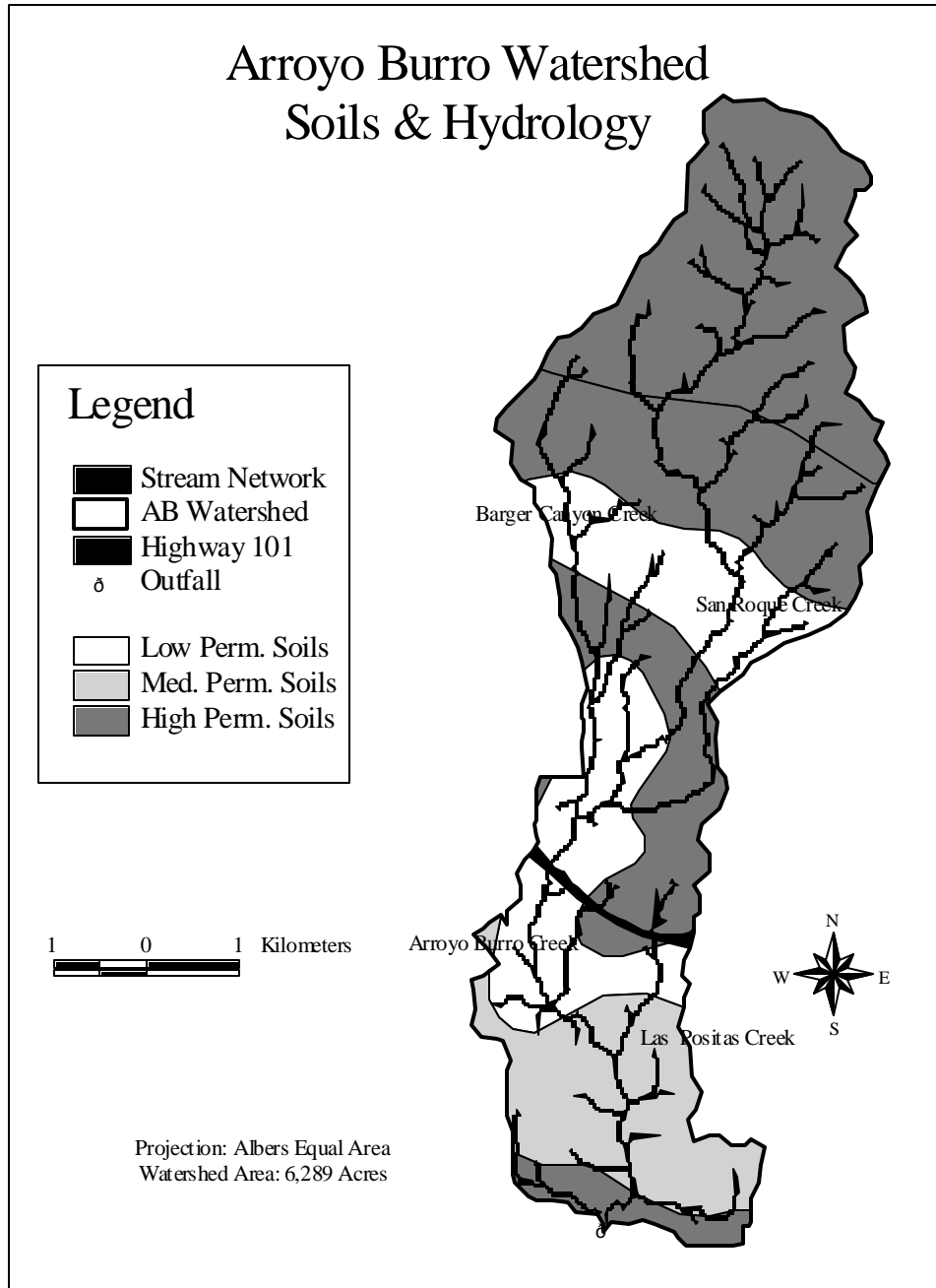
Appendix VI: Graphical illustration of the relationship between creek outfall bacterial counts and ocean bacterial counts

Bacterial Counts at Arroyo Burro Creek Outfall and in the Ocean
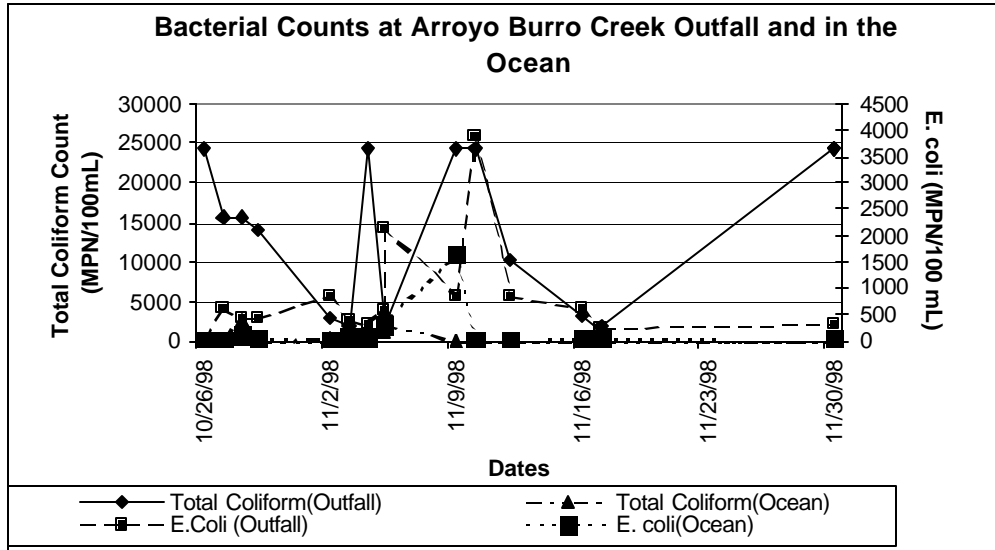
Appendix VII: Arroyo Burro Watershed Road Network & Land Use

Arroyo Burro Watershed
Road Network & Land Use

East Camino Cielo

Legend

AB Watershed
Road Network
Highway 101
Outfall

Commercial
Golf Course
Open Space
Residential
Natural Vegetation
Reservoir
Agriculture

Foothill Drive

State Street

Las Positas Road

1   0   1   Kilometers

N
W   E
S

Projection: Albers Equal Area
Watershed Area: 6,289 Acres
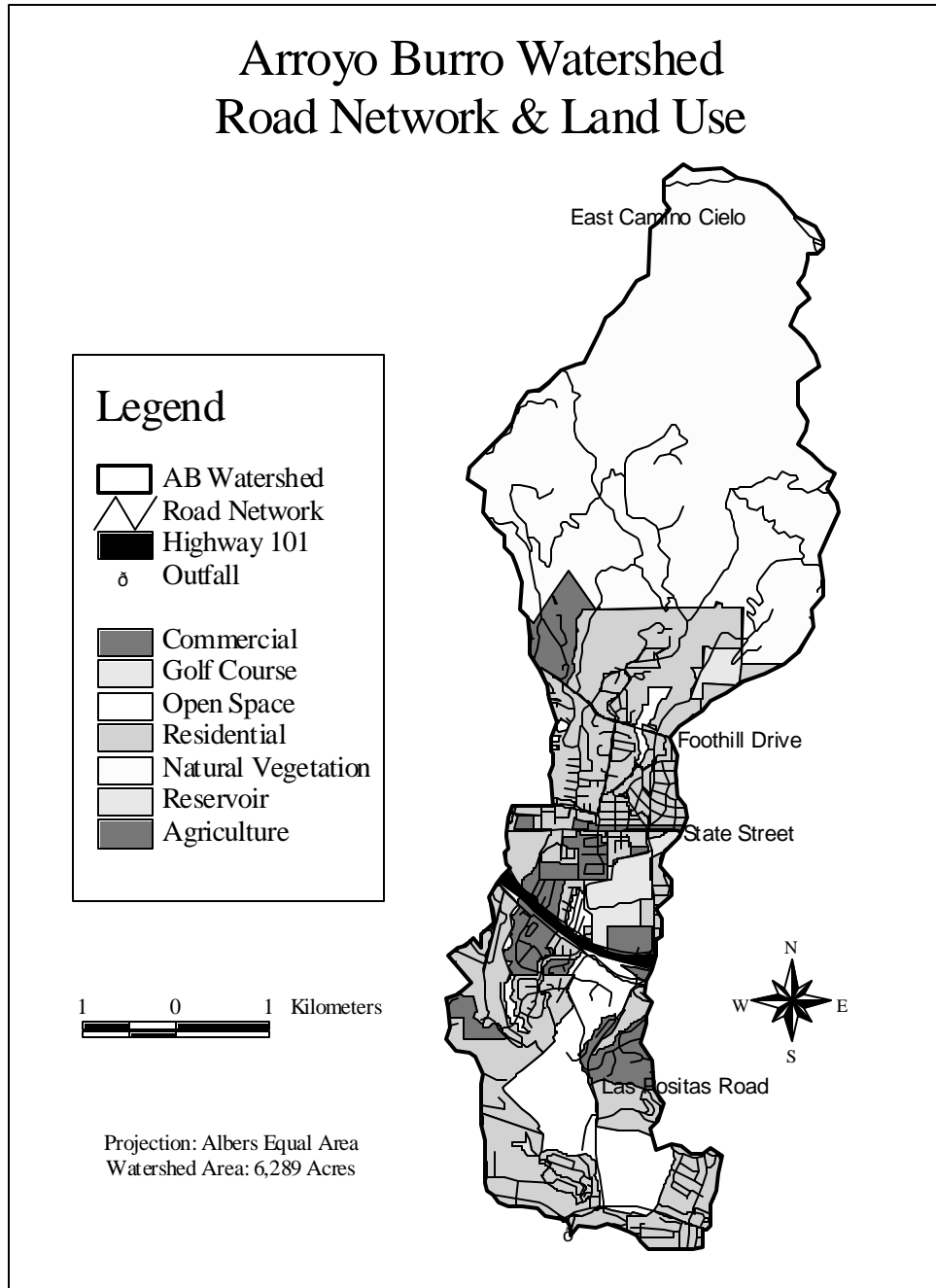
Arroyo Burro Watershed
Soils & Hydrology

Legend

Stream Network
AB Watershed
Highway 101
ᵭ Outfall

Low Perm. Soils
Med. Perm. Soils
High Perm. Soils

Barger Canyon Creek

San Roque Creek

Arroyo Burro Creek

Las Positas Creek

N
W   E
S

1   0   1 Kilometers

Projection: Albers Equal Area
Watershed Area: 6,289 Acres

Appendix IX: Runoff Curve Numbers (Average Watershed Condition)

Land Use Description		Curve Numbers for Hydrologic Soil Group			
		A	B	C	D
Fully developed urban areas[a] (vegetation established)					
Lawns, open spaces, parks, golf courses, cemeteries, etc.					
Good condition; grass cover on 75% or more of the area		39	61	74	80
Fair condition; grass cover on 50% to 75% of the area		49	69	79	84
Poor condition; grass cover on < 50% of the area		68	79	86	89
Paved parking lots, roofs, driveways, etc.		98	98	98	98
Streets and roads					
Paved with curbs and storm sewers		98	98	98	98
Gravel		76	85	89	91
Dirt		72	82	87	89
Paved with open ditches		83	89	92	93
	Avg. % impervious[b]				
Commercial and business areas	85	89	92	94	95
Industrial districts	72	81	88	91	93
Row houses, lot sizes 1/8 acre or less	65	77	85	90	92
Residential: average lot size					
¼ acre	38	61	75	83	87
1/3 acre	30	57	72	81	86
½ acre	25	54	70	80	85
1 acre	20	51	68	79	84
2 acre	12	46	65	77	82
	Hydrologic condition[c]				
Forestland-grass or orchards	Poor	55	73	82	86
	Fair	44	65	76	82
	Good	32	58	72	79
Brush	Poor	48	67	77	83
	Good	20	48	65	73
Forest-range					
Sage-grass	Poor		67	80	
	Fair		50	63	
	Good		35	48	

[a] For land uses with impervious areas, CN are computed assuming that 100% of runoff from impervious areas is directly connected to the drainage system.

[b] Includes paved streets.

[c] Poor = <30% ground cover density; Fair = 30-70% ground cover density; Good = > 70% ground cover density.

Source: McCuen 1989.

APPENDIX IX, p. 2: Adjustment of Curve Numbers for Condition II (Dry) and Condition III (Wet) Antecedent Soil Moisture Conditions.
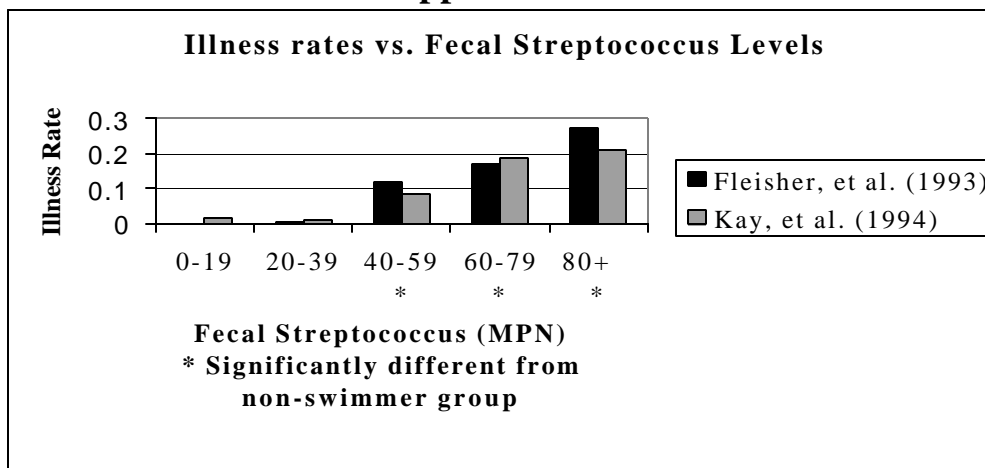
	Corresponding CN for:	
CN for		
Condition II	Condition I	Condition III
100	100	100
95	87	99
90	78	98
85	70	97
80	63	94
75	57	91
70	51	87
65	45	83
60	40	79
55	35	75
50	31	70
45	27	65
40	23	60
35	19	55
30	15	50
25	12	45
20	9	39
15	7	33
10	4	26
5	2	17

Source: McCuen 1989

Appendix X: SCS Rainfall Runoff Predictions for the Arroyo Burro and Arroyo Quemado Watersheds

Stormwater Runoff: Arroyo Burro and Arroyo Quemado (P=1")

Appendix XI

Illness rates vs. Fecal Streptococcus Levels

Fecal Streptococcus (MPN)
* Significantly different from
non-swimmer group

(a)

AB Enterococcus Levels Wet vs. Dry Season

(b)

Probabilities of Enterococcus Levels at Arroyo Quemado between 3/24/97-10/5/98
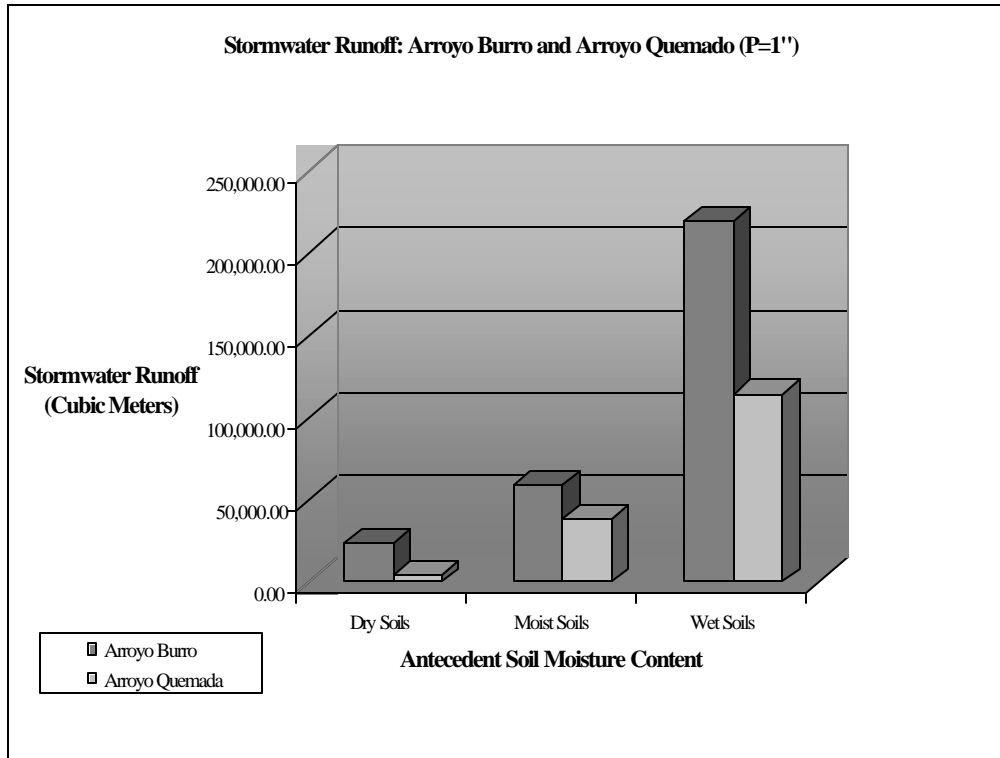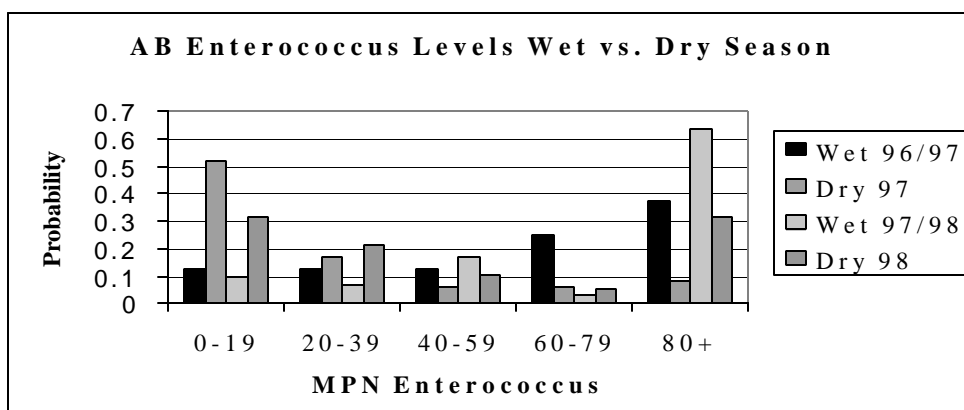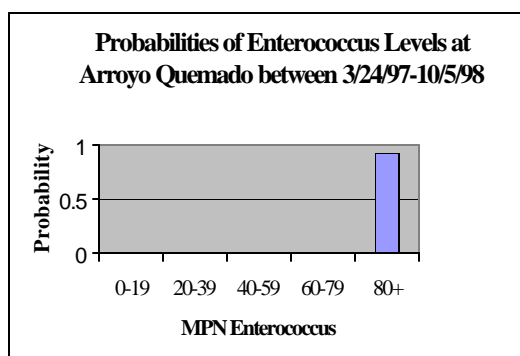
(c)

Figure (a) shows illness rate means minus non-swimmer illness rates in 20 MPN fecal streptococcus (FS) intervals for the two UK studies. Figure (b) shows the probability of seasonal enterococcus level occurrence at Arroyo Burro for a two-year period. Figure (c) shows the probability of enterococcus levels observed at Arroyo Quemado for 66 sampling dates between 3/24/97-10/5/98.

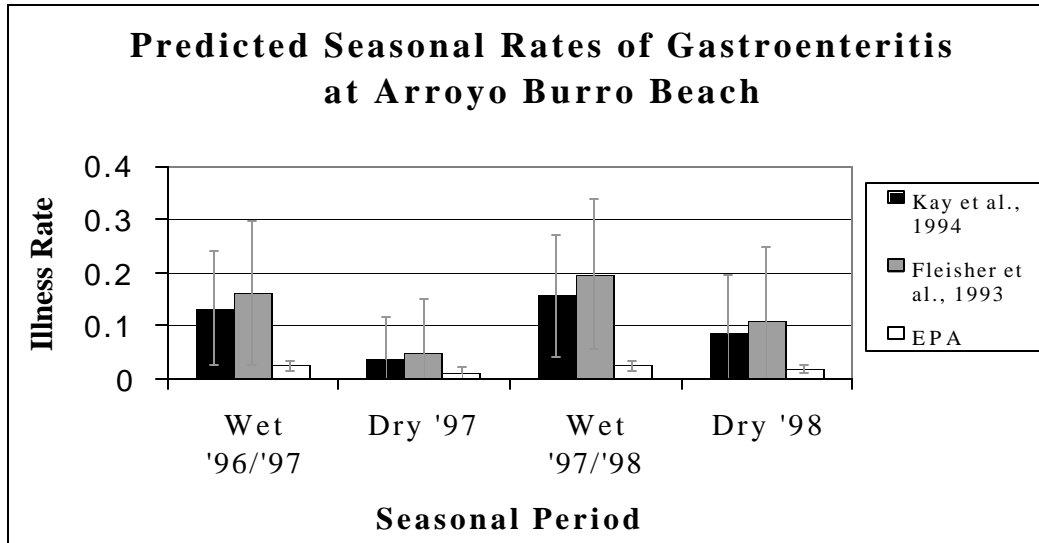# Appendix XII
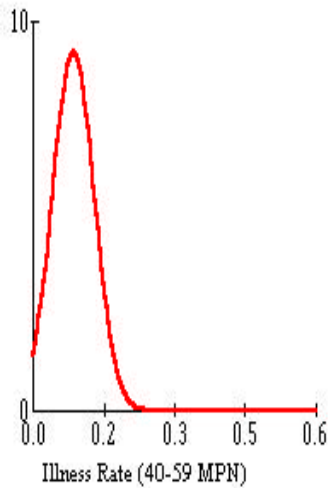
(a)
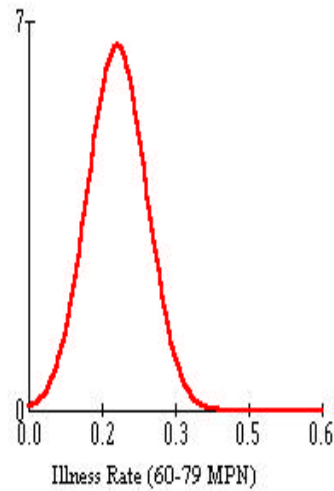
(b)
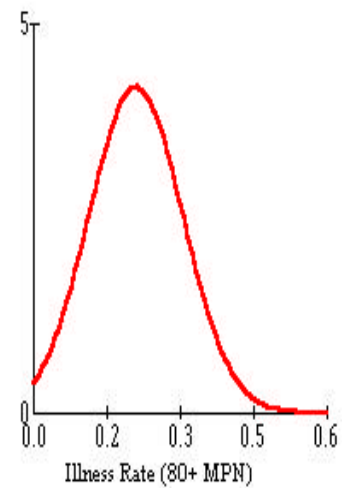
(c)
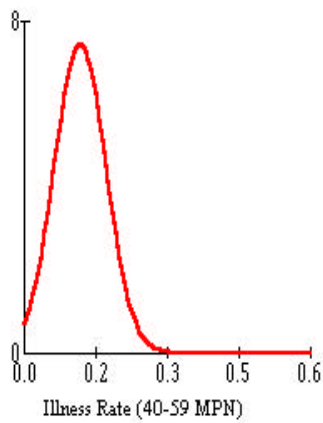
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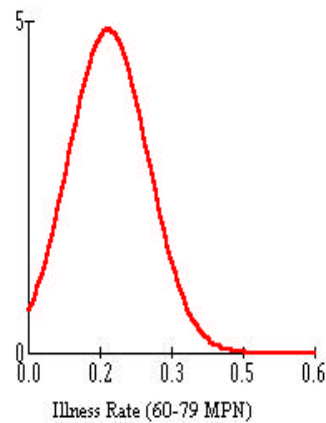
(e)
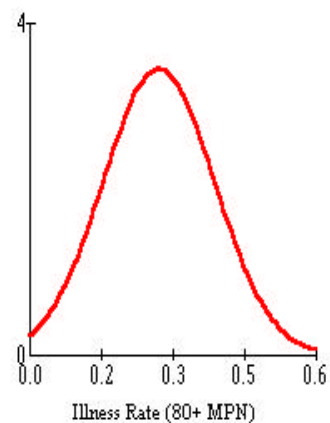
(f)

Each figure represents the uncertainty spread of the illness rate (x-axis). The illness rate is the fraction of the population expected to contract gastroenteritis. Figures a-c represent the illness rate uncertainty for fecal streptococcus (FS) level intervals: 40-59 MPN, 60-79 MPN, and 80+ MPN, respectively. These curves were calculated from data found in Fleisher, et al. (1993). Figures d-f represent the illness rate uncertainty for the same respective FS level intervals as above. These curves were calculated from data obtained from Kay, et al. (1994). These curves were used in combination with the site specific enterococcus data in a Monte Carlo simulation to
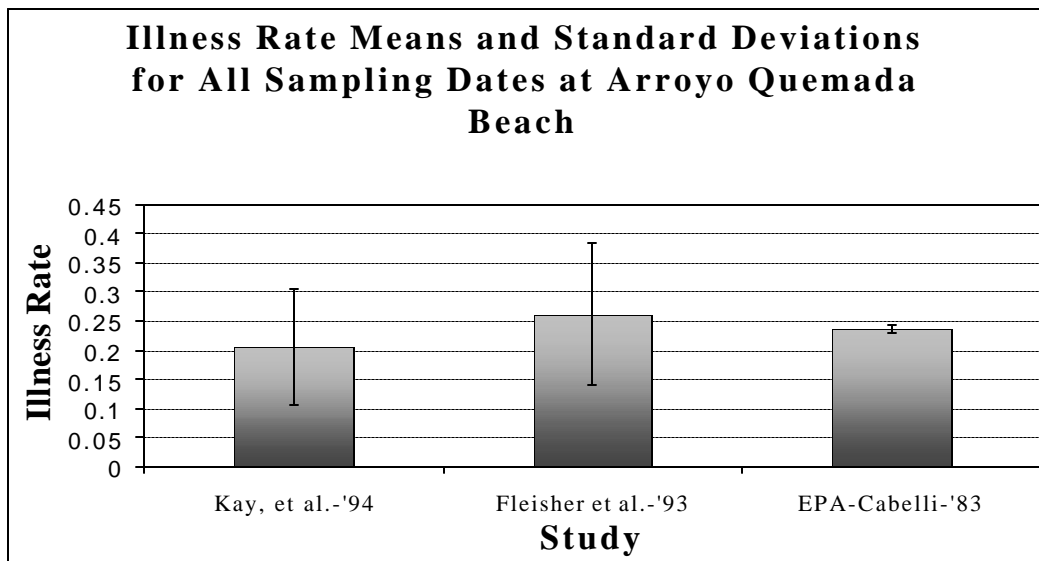
Appendix XIII
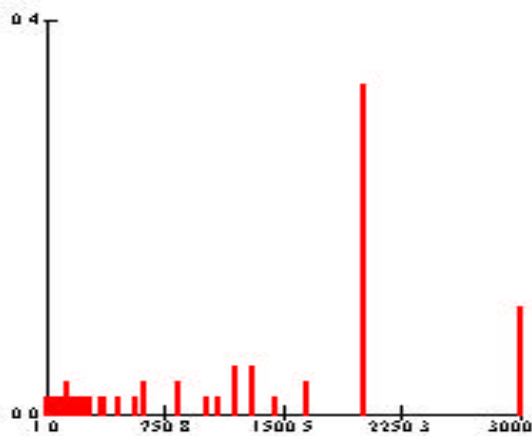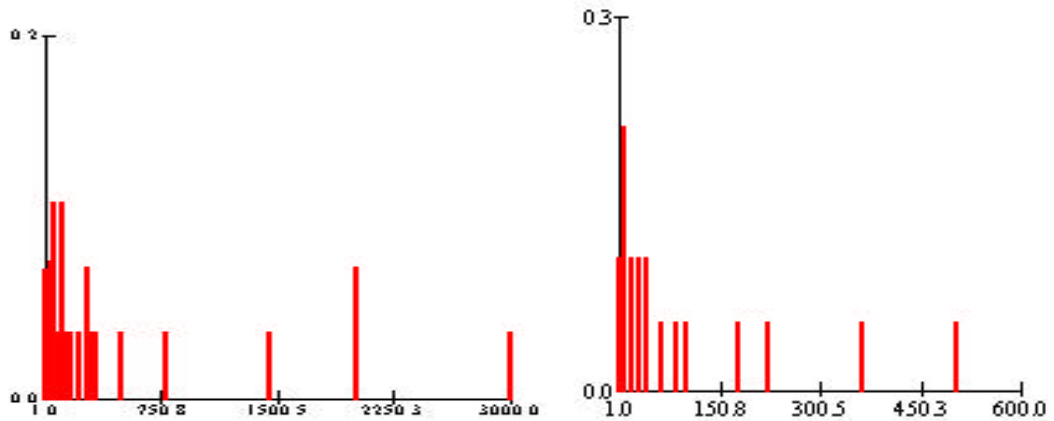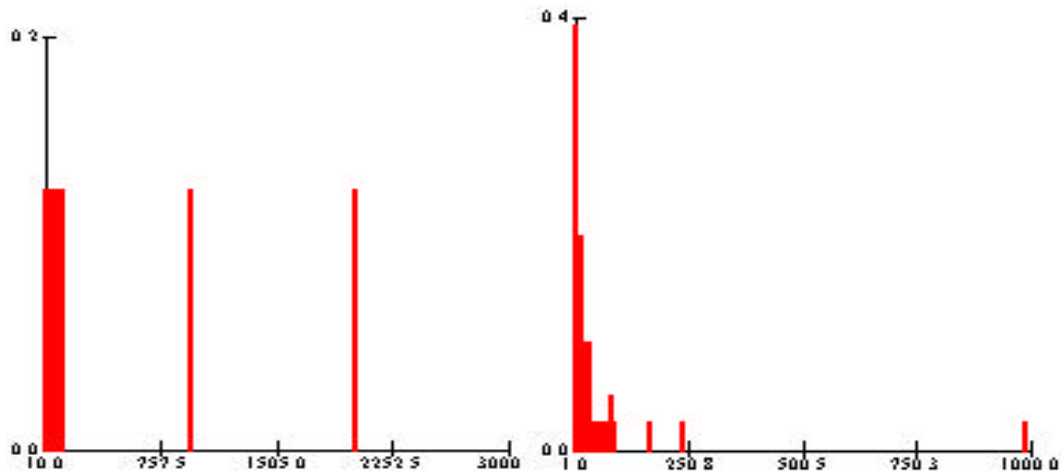
Each figure represents the discrete probability mass function (pmf) for enterococcus loading levels in MPN (x-axis) at a given location and period of time and probability of a given loading level (y-axis). These functions were used to determine the EPA dose-response values. In (a) we have the pmf for Arroyo Burro for the wet '96-'97 season. Fig. (b) shows the pmf for AB during the dry '97 season. Fig. (c) shows the pmf for Arroyo Burro during the wet '97-'98 season. Fig. (d) shows the pmf for Arroyo Burro during the dry '98 season. Finally, fig. (e) shows the pmf for Arroyo Quemado for all S.B. County sampling dates between 3/31/97-10/5/98.

Predicted Seasonal Rates of Gastroenteritis at Arroyo Burro Beach

(a)

Illness Rate Means and Standard Deviations for All Sampling Dates at Arroyo Quemada Beach

(b)

(a) This figure shows predicted illness rates (fraction of swimmer population ill) and
standard deviations calculated for seasonal periods at Arroyo Burro using the
three different dose-response relationships.

(b) This figure shows illness rate means and spreads at Arroyo Quemado for all three