
Technical Background Document: Microbial Risk Assessment and Fate and Transport Modeling of Aerosolized Microorganisms at Wastewater Land Application Facilities in Idaho



**Department of Environmental Quality
February 2006**

The cover photograph shows a pivot irrigated potato processing wastewater land treatment field in eastern Idaho.

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Abstract

This document provides technical and scientific background necessary for making quantitative assessments of risk to human health from microbial constituents in municipal and industrial wastewaters that are land applied. Both municipal and food processing wastewaters in Idaho contain various microbial constituents, which may have the potential to pose a risk to human health. To evaluate the relative risk of different land application practices, a quantitative microbial risk assessment methodology has been developed that uses microbial densities in air as critical input. The airborne transport pathway involves wastewater aerosolization, dispersion, deposition, and die-off. Irrigation droplet drift and aerosol transport are accounted for to predict microbial densities in air and deposition on surfaces downwind. The fate and transport approach is largely based on early EPA work (1982), with improvements made in aerosolization and dispersion/deposition modeling and in using the results to address human health impacts. A methodology has also been developed to provide an estimate of risk to public health given modeled microbial densities, type of receptor, mode of entry (ingestion or inhalation), and microorganism-specific characteristics. Preliminary model results suggest that drift and deposition of fine droplets at higher wind speeds may contribute to the risk of infection through ingestion of produce, a pathway not considered in the 1982 EPA guidance.

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FEBRUARY 2006

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Contents

Abstract	v
Acknowledgments	vii
List of Figures	x
List of Tables	xii
Executive Summary	xiii
Introduction	xiii
Approach	xiii
Recommendations to Enhance MIRA.....	xv
1. Introduction	1
1.0 Wastewater Land Treatment in the U.S. and Idaho	1
1.1 Wastewater Generation and Characteristics.....	2
1.2 Idaho Wastewater Land Application Rules and Guidance	2
1.3 Idaho Wastewater Land Application Practices.....	3
2. Aeromicrobiology	7
2.0 Occurrence of Microorganisms in Wastewater	9
2.1 Aerosolization of Wastewater Microorganisms	10
2.1.1 Aerosolization Process	10
2.1.2 Aerosolization Efficiency Estimates for Impact Sprinklers.....	11
2.2 Viability and Die-off of Wastewater Microorganisms.....	12
2.3 Bioaerosol Dispersion Field Studies	14
3. Quantifying Microorganism Fate and Transport: EPA Bio-Aerosol Modeling	17
3.0 Introduction	17
3.0.1 General Approach.....	17
3.0.2 Bio-Aerosol Prediction Model	17
3.1 Aerosol Source Strength (Q_s).....	18
3.1.1 Pathogen Content of Source Material (W).....	19
3.1.2 Material Application Rate (F)	20
3.1.3 Aerosolization Efficiency Factor (E)	20
3.1.4 Initial Organism Viability Reduction (or Impact) Factor (I).....	21
3.2 Microorganism Die-Off Factor (M_d)	22
3.3 Microbial Background Concentration (B)	24
3.4 Atmospheric Dispersion Factor (D_d).....	24
3.4.1 General Use of Dispersion/Deposition Modeling	24
3.4.2 Basis of Dispersion Models	25
3.4.3 Dispersion Models Applicable for Modeling Aerosols in Typical Conditions	26
3.4.4 Data Needs and Model Setup for Dispersion Modeling	29
4. Applicability of Existing Pathogen Dose-response Concepts to Wastewater	35
4.0 Introduction	35
4.1 Dose-response models vs. the Minimum Infective Dose Concept.....	36
4.2 Recommended Dose-response Models.....	38
5. Performing Microbial Risk Assessments	43
5.0 Recommendations – An Introduction	43
5.0.1 Hazard Identification	43
5.0.2 Dose-Response	43
5.0.3 Exposure Assessment.....	44
5.0.4 Risk Characterization.....	45

5.1	Estimating Infectivity (Dose-Response)	48
5.2	Estimating Exposure	48
5.3	Risk Characterization	52
5.4	Uncertainty Analysis	54
6.	Implementation of a Combined Fate, Transport and Risk Analysis Tool	57
6.0	Limitations of the Tool	57
6.1	Improvements Over the Original EPA Model	59
6.2	Aerosolization and Fine Droplet Fractions	59
6.2.1	<i>Equipment Specific Droplet Size Distributions</i>	59
6.2.2	<i>DRIFT02 Temperature Evaporation Model</i>	61
6.2.3	<i>Mist and Aerosolization Fractional Efficiencies Table</i>	63
6.3	Dispersion and Deposition Modeling	66
6.3.1	<i>Dispersion and Deposition Behavior vs Aerosol/droplet Size</i>	66
6.3.2	<i>Incorporating Refined Dispersion Modeling by Component</i>	69
6.3.3	<i>Worst Case Dispersion Outputs, With and Without Decay</i>	70
6.4	Risk Estimates	71
6.5	Model Calibration and Other Needs	71
6.6	Comparison of Modeled vs. Field Dispersion Studies	72
6.6.1	<i>Parker et al (1977) Measurements</i>	73
6.6.2	<i>Katzenelson and Teltsch (1976) Measurements</i>	73
6.6.3	<i>Conclusions from Limited Field Comparisons</i>	74
7.	Example Land Application Analysis	75
7.0	Hypothetical Pivot Operation	75
7.1	Aerosolization Efficiencies	76
7.2	ISC3 Dispersion Modeling for Pivot Components	76
7.3	Fate and Transport Parameters	77
7.4	Risk Parameters	77
7.5	Dispersion Modeling Results	78
7.6	Risk Analysis Results	82
7.6.1	<i>Results for Hypothetical configuration at 10⁶ CFU/100ml</i>	82
7.6.2	<i>Results for Reduced Wastewater Loading, 1,000 CFU/100 ml</i>	87
7.6.3	<i>Results for Alternative Sprinkler Nozzles, 10⁶ CFU/100ml</i>	89
7.7	Lessons Learned from Example Analysis	91
8.	Conclusions and Recommendations	93
	References	97
	Appendix A – Types of Idaho Wastewater Land Treatment Facilities	101
	Appendix B – Typical Wastewater Land Treatment Systems	105
	Appendix C – Wastewater Land Application Practices in Other Areas	109
	Appendix D – Microorganisms in Municipal and Food Processing Wastewaters in Idaho	111
	Index	115

List of Figures

Figure 1.	Irrigation drift generated by a movable big gun	4
Figure 2.	Pivot employing drag tubes for irrigation drift control at a potato processing wastewater land application site in eastern Idaho	4
Figure 3.	(Left) Linear move system employing drop tubes that put sprinkler heads closer to the ground	

at a potato processing wastewater land application site in southwestern Idaho. 5

Figure 4. (Right) Close up view of drop tube, off a linear move system with a sprinkler that generates coarse droplet distributions for irrigation drift control at a potato processing wastewater land application site in southwestern Idaho. 5

Figure 5. Vegetative buffer for both irrigation drift control and aesthetics at a potato processing wastewater land application site in southwestern Idaho. 6

Figure 6. DEQ Microbial risk conceptual model. 8

Figure 7 Effect of droplet size on aerosolization. 11

Figure 8. Aerosol efficiency estimate for rotating impact sprinklers. 21

Figure 9. Microorganisms die-off factor, M_d , as a function of distance from the source for various decay rates (λ). 24

Figure 10. Box model for estimating downwind concentration. 27

Figure 11. DEQ microbial risk assessment process. 58

Figure 12. Droplet distribution. 60

Figure 13. Relative quantities of droplets smaller than 0.2 and 0.1 mm for various sprinkler types. 61

Figure 14. Effect of temperature on percent droplet evaporated. 62

Figure 15. Effect of temperature and humidity on percent water loss from 0.1 and 0.2 mm droplets. 63

Figure 16. Air concentration versus droplet size with high wind speed. 67

Figure 17. Contribution to air concentration versus droplet size. 68

Figure 18. Percent contribution to surface deposition versus droplet size. 68

Figure 19. Dispersion factors for worst case day and night conditions. 71

Figure 20. Coliform aerosols downwind from a food processing facility. 73

Figure 21. Total coliform downwind from irrigation line. 74

Figure 22. Airborne microbe concentration versus atmospheric conditions. Low wind nighttime (F stability) conditions are the worst case for the inhalation pathway. 79

Figure 23. Microbe surface deposition versus atmospheric conditions. High wind conditions are the worst case for the deposition/produce consumption pathway. This shows the combined effect of airborne and deposited/ingested microbes on total dose. 79

Figure 24. Total dose versus atmospheric conditions. This shows the combined effect of airborne and deposited/ingested microbes on total dose. 80

Figure 25. Dose contributions for *E. coli*: F Stability, 1.0 mps. Inhalation is the predominant pathway under nighttime, very stable, low wind conditions. 81

Figure 26. Dose contributions for *E. coli*: D Stability, 10.0 mps. At high wind speeds, deposition on produce, followed by ingestion, contributes the greatest dose. 81

Figure 27. Dose contributions for *E. coli*: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions. 82

Figure 28. Daily risk contributions for *E. coli*: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions. 83

Figure 29. Annual risk contributions for *E. coli*: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions. 83

Figure 30. Dose contributions for *E. coli*: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and doses. 84

Figure 31. Daily risk contributions for *E. coli*: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and doses. 84

Figure 32. Annual risk contributions for *E. coli*: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and annual doses. 85

Figure 33. Variation in daily dose with distance, *E. coli*: D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings. 86

Figure 34. Variation in daily risk of infection, *E. coli*, D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings. 86

Figure 35. Variation in annual risk of infection, *E. coli*, D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings. 87

Figure 36. Dose contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml). 88

Figure 37. Daily risk contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml). 88

Figure 38. Annual risk contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml) 89

Figure 39. Dose contributions for *E. coli*: low-E nozzles at 10⁶ CFU/100 ml, D stability, 2.5 mps. 90

Figure 40. Daily risk contributions for *E. coli*: low-E nozzles at 10⁶ CFU/100 ml, D stability, 2.5 mps... 91

Figure 41. Annual risk contributions for *E. coli*: low-E nozzles at 10⁶ CFU/100 ml, D stability, 2.5 mps. 91

Figure 42. Solid set big gun irrigation system at a potato processing wastewater land application site in Eastern Idaho. 107

List of Tables

Table 1. Estimates of microorganism impact factor, I (from EPA 1982)^a 22

Table 2. Estimates of viability decay rate, λ (from EPA 1982)^a 23

Table 3. Minimum infective doses for selected bacterial and protozoan pathogens. 38

Table 4. Best-fit dose-response parameters from enteric pathogen ingestion studies. 40

Table 5. Aerosolization and fine droplet fractional efficiencies. 65

Table 6. Risk analysis parameters used in hypothetical example..... 78

Table 7. Total coliform counts (CFU/100 mL) in various wastewaters in Idaho. 112

Table 8. Fecal coliform counts (CFU/100 mL) of various wastewaters in Idaho. 112

Table 9. Microbial content of cheese processing wastewaters in Idaho. 113

Executive Summary

The Idaho Department of Environmental Quality (DEQ) developed this document to provide technical and scientific background for quantifying public health risks (if any) from microorganisms associated with land application of wastewater using spray irrigation. In addition, it describes specific methodologies that may be used to quantify human health risk from wastewater application. This document is not a how-to manual for risk assessment, but is rather the necessary first step toward developing practical tools for regulatory use.

Introduction

Wastewater land application involves distributing wastewater to the land surface so that the hydraulic load and nutrients in the wastewater may be beneficially re-used by an actively growing crop. Other constituents, such as organic material and inorganic salts having little agronomic significance, can be applied at rates such that they are assimilated and treated by the soil or effectively distributed in the system with minimal impact to the environment. There are currently more than 140 permitted wastewater land application facilities in Idaho; about 90 are municipal wastewater land treatment systems.

A concern surrounding the land application systems commonly used in Idaho is the prevention of irrigation *wind drift*, which includes droplet and aerosol drift. Wastewater land treatment facilities are often located in close proximity to dwellings, public parks and schools, rivers and streams, irrigation canals, roads, and other features that require special management of wastewater to protect health, safety, and the environment.

Because municipal and food processing wastewaters contain *microbial* constituents that could pose a risk to human health when land applied, the Idaho Department of Environmental Quality (DEQ) has developed a preliminary *microbial risk assessment* (MIRA) methodology to quantify this risk and to protect public health and safety.

Approach

Well-established methodologies used for air modeling and microbial risk assessment have been adapted to wastewater land treatment operations to help make site-specific determinations of 1) microbial densities, 2) potential health

risk, and 3) effects of modifications to irrigation systems and management on risk. The MIRA process described in this report involves first estimating pathogen concentrations in air and deposition on surfaces resulting from fine droplet drift and aerosol transport. Then the risk of an individual infection is estimated from exposure to pathogenic microorganisms via inhalation and ingestion pathways. The probability of infection is a function of the probability of inhaling or ingesting pathogens during an individual event, the number of application events, deposition rates on homegrown produce in the yard of a residential receptor, survival of pathogens on surfaces of fruits and vegetables, and frequency of produce consumption.

Using the MIRA process, risk can be calculated for each land application event, and annual and lifetime risks can then be calculated by combining risks from the aerosol and depositional pathways over the period of concern.

At this point in its development, the MIRA process has many limitations, including the following:

- 1) Aerosolization efficiencies and misting fractions extracted from Kincaid's database of spray drop measurements are static, rough approximations of droplet sizes that will actually aerosolize (dry to form a solid particle) rather than remaining as a mist droplet.
- 2) Dispersion treatment, though refined, still does not address, in an integrated manner, hourly changes in meteorology, evaporation rates, and droplet size.
- 3) Droplet and aerosol deposition on produce surfaces, though a common pathway in chemical risk assessment, has not been as extensively studied.

MIRA does not consider secondary transmission, which refers to an individual being infected and then transferring the infection to another individual. It could be assumed that if the risk of primary infection from exposure to wastewater is maintained at an acceptably low level, secondary transmission will not increase risk to an unacceptable level. This assumption may not be entirely accurate, and it is recommended that this factor be considered in the further development of these risk assessment methodologies.

Recommendations to Enhance MIRA

To supplement and enhance MIRA, the following actions are recommended:

1. Rigorously characterize typical food processing, industrial, and municipal wastewaters in Idaho for indicator species as well as for wastewater-specific pathogenic microorganisms. This information will provide the microbial source term for drift/deposition modeling.
2. Characterize diverted surface irrigation waters in representative irrigation districts for indicator species as well as for specific pathogenic microorganisms. Modeling drift/deposition of microbial constituents from irrigated agriculture will provide an important reference point between common and longstanding irrigation practices, expected exposures, and wastewater land application practice.
3. Develop an automated modeling tool to characterize changing microbial deposition and microbial air densities along the flow path for both aerosols and the trajectories of larger particles. Exposure to microorganisms in these various and transitioning states can be summed and utilized in microbial risk assessments.
4. Compare computer model results with field measurements. For budgetary reasons, this activity should be limited to characterizing deposition onto deposition plates – an inexpensive analysis – to characterize both deposition and microbial densities indirectly. Depending upon the correlations obtained in field studies, modeling parameters can be modified and the model calibrated to actual Idaho field conditions.
5. Develop stand-alone, pre-run model output tables that the regulated community can use to design and operate wastewater land treatment systems to minimize microbial risk.
6. Develop example calculations to further illustrate how the equations are used, which units are used, how units cancel, and how output from calculations are subsequently used as input into successive calculations.
7. Apply these recommendations to site-specific permitting circumstances, so that other modifications and adaptations may suggest themselves for inclusion in revisions to these recommendations.

FEBRUARY 2006

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1. Introduction

Municipal and food processing wastewaters contain *microbial* constituents that could pose a risk to human health when such wastewaters are land applied; to quantify this risk, the Idaho Department of Environmental Quality (DEQ) has developed a preliminary *microbial risk assessment* (MIRA) methodology for protecting public health and safety at regulated wastewater land application facilities.

This document provides technical and scientific background necessary for making quantitative assessments of risk to human health from microbial constituents in municipal and industrial wastewaters that are land applied. This document is not a how-to manual for risk assessment, but is rather the necessary first step toward developing practical tools for regulatory use.

1.0 Wastewater Land Treatment in the U.S. and Idaho

Land application of wastewater has long been recognized as an effective method to treat wastewater generated by a variety of industries and municipalities. Crites et al. (2000) provides a brief history of wastewater land treatment, including practices in Europe and the United States.

Of particular significance is a series of reports on wastewater treatment authored in the 1890s by George Rafters of the U.S. Geological Survey, who concluded that “sewage farming” is “not prejudicial to health.” At the time when Rafters was writing, “most of the 143 sewage treatment facilities in the United States and Canada ... were land treatment systems ...”. After that time, wastewater land treatment began to decline due to several factors, the most important being “the development of the germ theory for disease transmission, with the use of chlorine as a disinfectant making it ‘safe’ to discharge partially treated sewage to waterways.” (Crites et al., 2000 p. 3).

Later, as the environmental and health impacts of discharging partially treated sewage to waterways became increasingly recognized, wastewater land application began once again to be a serious alternative for consideration. Since the passage of the Clean Water Act of 1972, wastewater land application has come back into widespread practice.

In Idaho, there are currently more than 140 permitted wastewater land application facilities. About 90 of these are municipal wastewater land

treatment systems. The remainder consists of industrial systems, including potato, cheese, sugar beet, and meat processing facilities. Wastewater volumes generated by these permittees range from a few million gallons per year to over a billion gallons per year for the largest industrial processors.

Many Idaho residents remember the condition of the Snake River when municipal, and especially food processing, facilities discharged to the river, and how dramatically the river water quality changed when these facilities converted to land treatment. Ground water contamination and nuisance odor conditions did however result at certain facilities as a result of wastewater land application. Those conditions were driving forces in developing a land application permit program in Idaho.

See Section 1.2 for further discussion of Idaho's Wastewater Land Application Program rules and guidance.

1.1 Wastewater Generation and Characteristics

Each wastewater land treatment facility is unique in terms of wastewater streams, hydrogeology, soils, climate, season of application, and other factors, with wastewater quality and quantity varying significantly from one facility to another. Wastewater characteristics also vary with industrial process, particularly with the product being made (e.g., potato flakes versus diced potatoes), type and extent of treatment, storage and detention times, and physical state, age, and quantity of vegetative material being processed, among other factors.

For further background information on wastewater generation processes and characteristics of industrial and municipal wastewater land application facilities in Idaho, see Appendix A, page 101.

1.2 Idaho Wastewater Land Application Rules and Guidance

The Department of Environmental Quality (DEQ) is responsible for protecting public health and safety in the implementation of the rules it administers. Three agency documents are of special importance in wastewater land application in the state of Idaho:

- To protect public health, safety, and the waters of the state, Idaho's *Wastewater-Land Application Permit Rules* (IDAPA 58.01.17) were first promulgated in 1988. These regulations are primarily procedural, outlining

regulatory steps and timeframes for applying for and being issued a wastewater land application permit.

- The companion *Guidelines for Land Application of Municipal and Industrial Wastewater*, were issued by DEQ in March 1988 (DEQ 1988).
- The Handbook for Land Application of Municipal and Industrial Wastewater (DEQ 1996) and the updated version of this document (October 20, 2004) constitute the latest program guidance available.

Together, these regulations and guidelines help establish parameters for writing wastewater land application permits that protect surface and ground water quality, protect public health and safety, and meet the treatment needs of the wastewater generator.

1.3 Idaho Wastewater Land Application Practices

Wastewater land application involves distributing wastewater to the land surface so that the hydraulic load and nutrients in the wastewater may be beneficially re-used by an actively growing crop. Other constituents, such as organic material and inorganic salts having little agronomic significance, can be applied at rates such that they are assimilated and treated by the soil or effectively distributed in the system with minimal impact to the environment. Microorganisms encountering the soil are, in most cases, effectively filtered in the soil matrix, and die off at various rates (EPA 1992, Table 4).

A concern surrounding wastewater land application practices, especially as they pertain to slow rate systems commonly used in Idaho¹, is the prevention of irrigation wind drift which includes droplet and aerosol drift. Wastewater land treatment facilities are often located in close proximity to dwellings, public parks and schools, rivers and streams, irrigation canals, roads, and other features that require special management of wastewater to protect health, safety, and the environment².

Certain irrigation methods, such as big gun sprinklers, can generate significant amounts of droplet and aerosol drift (Figure 1).

¹ For a more detailed discussion of typical wastewater land application treatment systems, see Appendix B.

² For a review of wastewater land application practices in the U.S. and worldwide, see Appendix C.



Figure 1. Irrigation drift generated by a movable big gun.



Figure 2. Pivot employing drag tubes for irrigation drift control at a potato processing wastewater land application site in eastern Idaho.

To prevent droplet and aerosol drift of applied wastewater, certain facilities are using drag tubes on pivots and linear moves instead of sprinklers (see Figure 2). A typical pivot configuration for drag tubes consists of mounting metal drop tubes at regular intervals along the pivot. These drop tubes extend above the land surface about four to five feet. Attached to the end of the drop tube is a manifold to distribute water flow between three to five outlets. The flexible drag tubes are then attached to the outlets on the manifold and are sized so that about two feet of tube can drag parallel to the ground surface. Linear move systems, mentioned above, have also been successfully configured with drag tubes.

Other facilities are mounting sprinklers closer to the ground surface (Figure 3) to achieve less drift. Also employed are sprinklers that generate coarser droplet sizes (Figure 4). Coarser droplet sizes greatly minimize irrigation droplet and aerosol drift.



Figure 3. (Left) Linear move system employing drop tubes that put sprinkler heads closer to the ground at a potato processing wastewater land application site in southwestern Idaho.

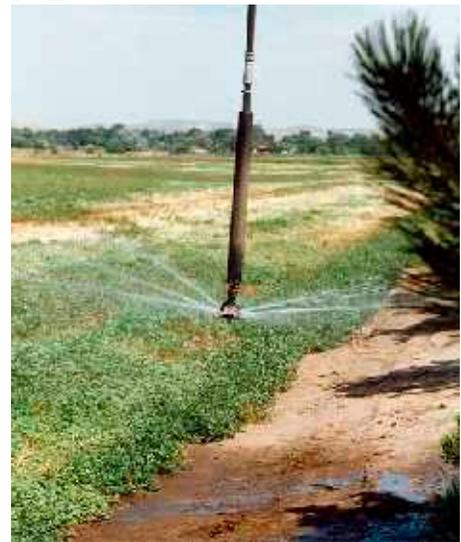


Figure 4. (Right) Close up view of drop tube, off a linear move system with a sprinkler that generates coarse droplet distributions for irrigation drift control at a potato processing wastewater land application site in southwestern Idaho.

Vegetative buffers are also utilized to mitigate irrigation drift (Spendlove et al. 1980) by intercepting drift and creating more turbulence and mixing of aerosols to decrease their concentration. Vegetative buffers also serve to intercept droplets and aerosols. Figure 5 shows a mature vegetative buffer.



Figure 5. Vegetative buffer for both irrigation drift control and aesthetics at a potato processing wastewater land application site in southwestern Idaho.

2. Aeromicrobiology

Aeromicrobiology is the study of aerosolization, aerial transmission, and deposition of biological material. A brief introduction to aeromicrobiology is provided in Maier et al., 2000, and is summarized as follows:

- Bioaerosols consist of fungi, bacteria, and viruses, often in combination with inorganic particulates and/or moisture. Bioaerosol particles are considered small if their diameters are less than 0.1 micrometer (μm). Moderate sized particles are between 0.1 and 2.0 μm in diameter. Coarse particles are from 2.0 to 100 μm in diameter. Bioaerosols are largely (though not exclusively) transported in the boundary surface layer of the atmosphere, extending about 100 m from the ground surface. The upper part of this layer is always turbulent while that close to the ground surface is relatively still.
- The aeromicrobiologic transport pathway begins with bioaerosols being launched into the air. For purposes of this document, the means of launching is an irrigation system. Once launched, bioaerosols are then transported, experiencing diffusion and dispersion. Transport is characterized in terms of time in transit and distance traveled.
- Deposition occurs through gravitational settling, rain, electrostatic forces, surface impaction, and downward turbulent diffusion. As discussed in Section 3.0, microorganism viability in bioaerosols is influenced by relative humidity, temperature, radiation (particularly ultraviolet and X-rays), oxygen toxicity, and incompletely understood '*open air factors*'.

Working together, these effects define a pathway for microbial transport, human exposure, and risk of infection (Figure 6) from which it is possible to estimate daily and annual risk due to inhalation and ingestion of pathogens originating from wastewater land application.

Conceptual Model of Human Infection from Wastewater Land Application

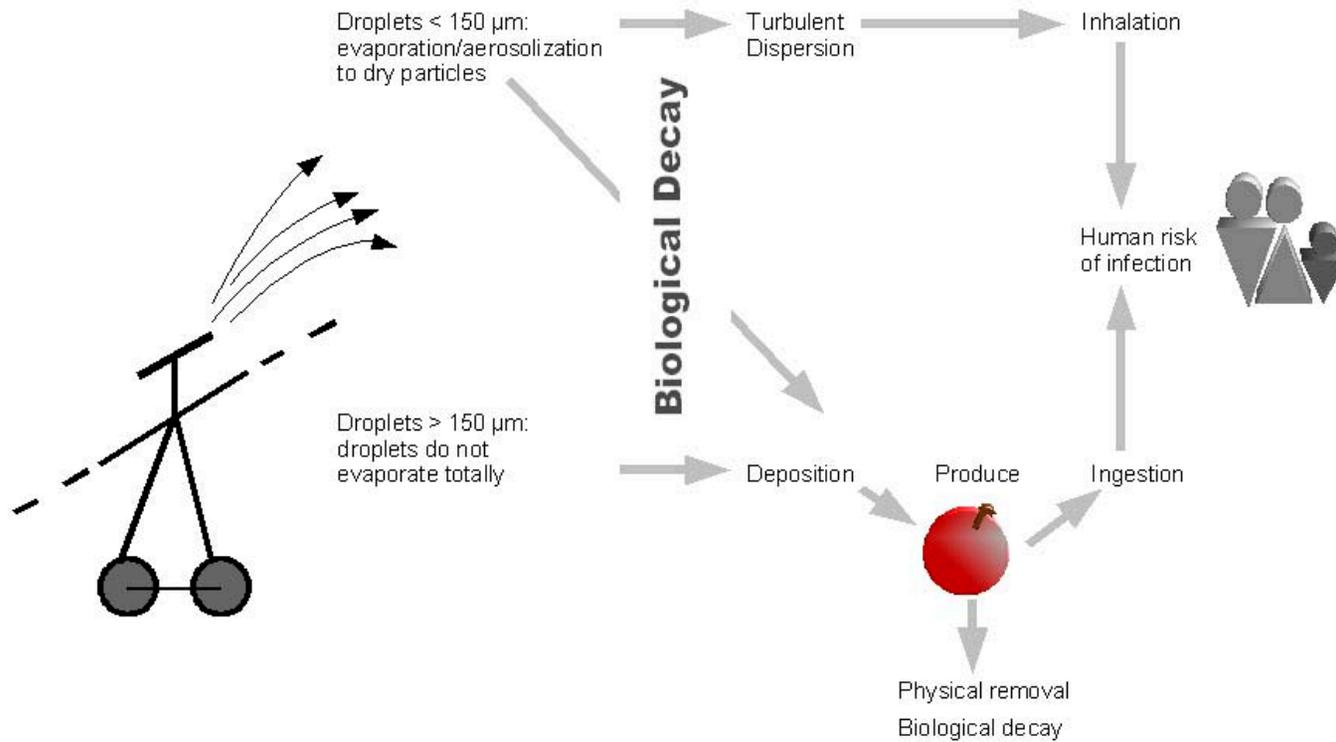


Figure 6. DEQ Microbial risk conceptual model.

2.0 Occurrence of Microorganisms in Wastewater

Microorganisms are ubiquitous in the environment, and industrial and municipal wastewaters are no exception. Where there is substrate (food), an aqueous environment, appropriate temperatures, time, and microorganisms, there will almost certainly be microbial growth. Microbial characteristics of land applied wastewater prior to land application can be obtained from existing, albeit limited, data sets, or from site-specific sampling and analysis. Irrigation drift and aerosolized particles resulting from spray irrigation of wastewater are also characterized through irrigation drift and aerosol modeling. Microbial and aerosol drift characterization is necessary to determine the magnitude and type of air-borne microorganisms that are viable and have the capacity to infect a human receptor.

Microbial content of municipal wastewaters is due largely to those enteric microorganisms occurring in fecal matter. Lists of typical pathogenic organisms occurring in municipal wastewaters, respective numbers of organisms, and survival times in the environment can be found in EPA (1992; Tables 1 and 3), Tchobanoglous and Burton (1991; Table 3-18), FAO (1992; Section 1.2, Tables 4 and 5), and Gerba and Smith (2004; Tables 5 and 6).

Limited data sets for total and fecal coliform content of Idaho municipal and food processing wastewaters permitted for land application are found and discussed in Appendix D, page 111. Also in Appendix D are limited microbial data for cheese processing wastewaters.

Due to the limited nature of the above referenced data sets, a rigorous characterization of typical food processing, industrial, and municipal wastewaters in Idaho for wastewater-specific pathogenic microorganisms, as well as indicator species, as applicable, is recommended to have better microbial source terms for drift/deposition modeling. (See Section 3, page 17, for more on bioaerosol modeling.)

Also, characterizing diverted surface irrigation waters in representative irrigation districts for specific pathogenic microorganisms and indicator species, as applicable, is recommended to provide an important reference point between common and longstanding agricultural irrigation practices, expected exposures, and wastewater land application practice.

2.1 Aerosolization of Wastewater Microorganisms

Aerosolization refers to the process in which fine spray droplets containing wastewater microorganisms evaporate to dryness or near dryness, leaving a much smaller solid or semi-solid particle or bio-aerosol. When this occurs, the smaller bioaerosol may travel much farther than the original droplet and may be an important component of the total microbial risk.

2.1.1 Aerosolization Process

Any spray application system applies wastewater to the land surface by breaking the wastewater stream up into droplets. Depending on the pressure and nozzle configuration, the droplets may range from very fine ($< 100 \mu\text{m}$ or 0.1mm) to large ($> 1 \text{ mm}$ or $1,000\mu\text{m}$). When the atmospheric humidity is less than 100%, evaporation of moisture from the droplets occurs. The evaporation rate increases with lower humidities and higher temperatures.

As droplet size becomes smaller, the surface-area-to-water-volume ratio increases rapidly. As a result, droplets larger than about $200 \mu\text{m}$ don't evaporate appreciably, while droplets smaller than $100 \mu\text{m}$ in diameter evaporate rapidly. If a droplet evaporates totally before it strikes the ground or any other surface, it becomes aerosolized and disperses further downwind before being removed by surface deposition.

Whenever the evaporation time is less than the fall time aerosolization occurs. For fall distances appropriate for many spray irrigation systems, (10 – 15 ft), droplets less than $150 \mu\text{m}$ are generally expected to become aerosolized under moderate humidity conditions. This behavior can be seen in Figure 7.

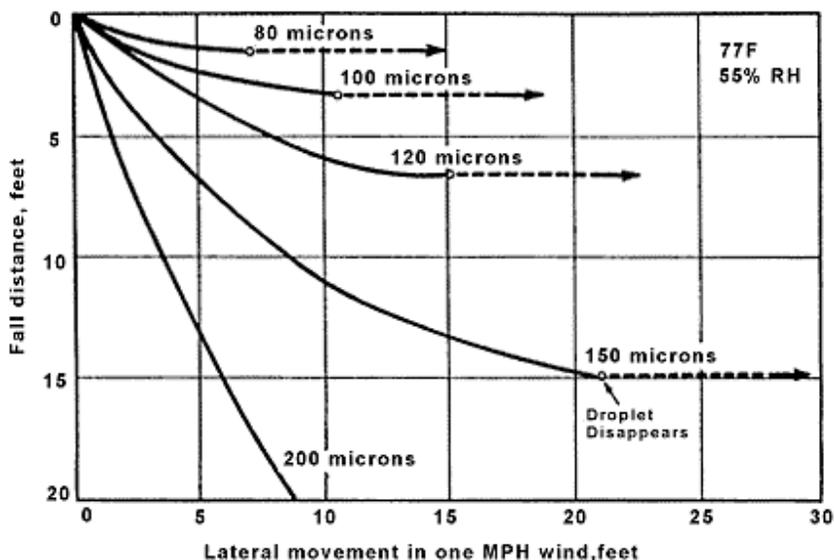


Figure 7 Effect of droplet size on aerosolization.

2.1.2 Aerosolization Efficiency Estimates for Impact Sprinklers

Because aerosolized microbes can travel much farther than the original droplets, the fraction of the wastewater spray flow that becomes aerosolized is an important factor in estimating microbial risk.

The fraction of total wastewater that is sprayed into the air and that leaves the vicinity of the irrigation system as dry or semi-dry aerosols rather than droplets is defined as Aerosolization Efficiency (EPA, 1982). The Aerosolization Efficiency (E) has been determined under a wide variety of conditions by injecting a fluorescent dye into the irrigation system, sampling for it downwind, and making parallel dispersion modeling predictions assuming all the irrigation water is aerosolized. The ratio between the measured and predicted dye concentrations is the Aerosolization Efficiency.

EPA (1982) reported median aerosolization efficiency measurements from rotating impact sprinklers at three different sites to be 0.0033 (with a range of approximately 0.001 to 0.002). This means that 0.33 percent of the total sprayed wastewater flow leaves the area as aerosol.

EPA used aerosolization data from one of their three measurement sites to develop a relationship for E that reflects the influence of wind speed, solar radiation and temperature for rotating impact-type sprinklers. This relationship is described further in Section 3.1.3, page 20.

2.2 Viability and Die-off of Wastewater Microorganisms

Survivability and viability differ widely among microorganisms in sprayed bioaerosols. *Escherichia coli* (*E. coli*), the most frequently monitored coliform bacteria in wastewater, generally have an extremely short life span in aerosol form (Poon 1966). Sorber and Guter (1975) reported that atmospheric bacterial die-off is geometric in nature with the majority of the organisms dying within 3 seconds. The remaining resistant bacteria continue to die at a decreasing rate with time. A study by the United States Environmental Protection Agency (EPA, 1982) showed that the median viability decay rates (λ) for total coliform and fecal coliform in aerosols from spray irrigation of wastewater are -0.0332 sec^{-1} and -0.023 sec^{-1} , respectively. These two types of microorganisms died off fastest among those tested. Die-off rates of fecal streptococci and enterovirus were too slow to be detected.

Physical characteristics of the aerosol and environmental factors are primary parameters for the survival and viability of microorganisms within an aerosol (Brooks et al. 2004). Size, shape, chemical composition, moisture content and density of the aerosol strongly influence longevity as well as transport of microorganisms. The extent of inactivation due to these factors also depends on the type of organisms. In general, *gram-positive* bacteria—those stained blue or violet (e.g. staphylococci and streptococci) by the gram-staining process—survive better in an aerosolized state than *gram-negative* bacteria (e.g. salmonella). Moreover, spray irrigation devices with high aerosolization efficiencies generate smaller bioaerosols, which cause microorganisms to be more dispersed and vulnerable to die-off. Aggregated microorganisms or microorganisms associated with particles are generally more protected from inactivation and settle faster from the air. Bioaerosols with high organic contents are often protective to the microorganisms. Pathogens (disease-causing microorganisms) with the ability to form spores and cysts are generally much more resistant in the environment and live longer than vegetative cells. The better survivals are due to the protection provided by spores and the outside shell of the cysts. Also, viruses survive better when enveloped in lipid.

The high die-off rates in aerosols may be due to aerosolization forces and the sampling stresses, which adversely impact the ability of microorganisms to be cultured on growth media (i.e., *culturability*) as well as the viability of the organisms. During aerosolization of spray-irrigated wastewater, the bacteria may be broken to pieces by the aerosolization forces and sampling stress.

These pieces remain viable yet lose the ability to be cultured (Heidelberg et al., 1997). Therefore, resulting die-off and viability estimation may underestimate the actual risk present in the aerosols. Currently, molecular techniques, such as polymerase chain reaction (PCR) are recognized to identify bioaerosol microorganisms both qualitatively and quantitatively.

Important environmental factors include relative humidity, temperature, ultraviolet radiation, and the method of aerosol generation (Brooks et al. 2004). Cox (1987) has suggested that relative humidity and water content are the most important environmental factors influencing bioaerosol stability. In general, increased relative humidity tends to increase the water sorption of the organisms and, therefore, protects them from desiccation and ultraviolet-induced inactivation. However, the effect of relative humidity on survivability of organisms may interact with the toxic effect of oxygen. Cox's early work (1966) found that aerosolized *E. coli* in a nitrogen atmosphere exhibited 100% survival during the conditions of moderate relative humidity (40-50%) with enhanced decay observed when relative humidity is above 80%, therefore, published high die-off rates of *E. coli* at low relative humidities may not be due to the single effect of desiccation but from the toxic effect of oxygen as well.

High temperature tends to promote desiccation, which is unfavorable to growth of bacteria such as *E. coli*. Ultraviolet radiation contributes significantly to inactivation of bioaerosols. The ultraviolet and/or visible rays inactivate the microorganisms by deforming DNA and damaging normal cellular functions. Teltsch et al. (1980a) showed that die-off rate of aerosolized *E. coli* from sprinkler application of wastewater differed between early morning and afternoon. Decaying of the aerosolized *E. coli* is faster in the afternoon at 0.066 sec^{-1} comparing with $8.8 \times 10^{-3} \text{ sec}^{-1}$ in the early morning. These studies assumed first-order die-off kinetics.

Teltsch et al. (1980a) indicated that die-off rates of aerosolized microorganisms from spray irrigation systems are much higher than those found in natural waters, such as streams, oceans, and sewage plants. Other literature suggests that the typical die-off rates range from 0.29 to 0.43 day^{-1} (3.3×10^{-6} to $4.9 \times 10^{-6} \text{ sec}^{-1}$) and 0.22 to 0.34 day^{-1} (2.5×10^{-6} to $3.9 \times 10^{-6} \text{ sec}^{-1}$) for total coliform and fecal coliform respectively in natural waters (Easton 1999, Nasser 2003).

2.3 Bioaerosol Dispersion Field Studies

Significant effort has been expended to characterize bioaerosols and quantify background and downwind microbial densities (concentrations) in biosolids and municipal wastewater spray application. There are several literature reviews that summarize the present state of knowledge (Pillai and Ricke 2002, McEwen, 1997; Forcier 2002, EPA 1992, and Sorber and Sagik 1978). Stetzenbach (2002) has compiled an exhaustive table of microbial aerosol concentrations for different sources including wastewater treatment, and accompanying references. Bioaerosols from wastewater spray irrigation are discussed here.

Bausum, et al. (1982) conducted a major study of municipal wastewater bioaerosols (standard plate count and coliphage f2). Type of samplers used greatly influenced net aerosol strength (microbial densities) which was measured. Unchlorinated wastewater applied during daylight hours resulted in somewhat lower net bacterial aerosol strength than when applied at dusk or night. Although highly variable, net bacterial and coliphage (virus) aerosol strength decreased with distance from the source. Spray irrigated chlorinated wastewater resulted in bacterial aerosol densities near background levels.

Parker et al. (1977) studied bioaerosols from potato processing wastewater spray fields. Coliform bioaerosols from the spray field were collected 392 m from the source. Aerosolized enteric bacteria (coliform and salmonella) from a spray irrigated field near Kibbutz Tsorah, Israel were studied by Katzenelson and Teltsch (1976). Aerosolized coliform and salmonella bacteria were found 350 m and 60 m downwind respectively.

Johnson et al. (1980) conducted a major study of bioaerosols from a wastewater spray irrigation field in Pleasanton, California. Bioaerosol densities of several enteric species were determined at several downwind locations. Total and fecal coliform, coliphage, fecal streptococci, pseudomonas, mycobacteria and enterovirus were found in bioaerosols downwind at levels above background. As seen in data from Bausum et al. (1982), net aerosol strengths of all species decrease over distance.

Brenner et al. (1988) observed heterotrophic (standard) plate count aerosolized bacteria ranging from 86 to 7,143 Colony Forming Units per cubic meter (CFU/m³) downwind of a wastewater sprayfield in Muskegon, Michigan. Animal viruses were present in wastewater but were not recovered in aerosol sampling, possibly due to insufficient numbers in the source water.

Teltsch and Katzenelson (1978) studied sewage effluent spray irrigation near Ein Kerem, Israel. They found a positive correlation between relative humidity and aerosolized bacteria (total coliform and marker *E. coli*) and a negative correlation between solar radiation and aerosolized bacteria levels. They found a ten-fold increase in aerosolized bacteria during night irrigation. Echovirus 7 was present in samples 40 m downwind of the source.

Teltsch et al. (1980a) studied aerosol levels of salmonellae, total coliforms, and enterovirus near Kibbutz Tsorah, Israel. Total coliforms, salmonellae, and enterovirus were detected up to 200 m, 100 m, and 40 m from the source respectively. Ratios of salmonellae/total coliform and enterovirus/total coliform were examined for both aerosols and wastewater. Both ratios were much greater for aerosols compared with wastewater, indicating greater rate of coliform die-off compared to salmonellae and enterovirus. The authors raised questions as to the utility of total coliform as an indicator species for aerosols.

Camann et al. (1988) found elevated levels of fecal coliform, fecal streptococci, and mycobacteria 200 m downwind of a municipal wastewater land application field in Lubbock, Texas.

FEBRUARY 2006

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3. Quantifying Microorganism Fate and Transport: EPA Bio-Aerosol Modeling

3.0 Introduction

Mathematical aerosol dispersion models can be used to estimate the concentration of bacteria in air at specific points in time and space. Modeled microbial concentrations in air are a function of specific location and strength of the source and environmental factors affecting the aerosol generation, organism die-off, and atmospheric dispersion and deposition. These models can be configured for screening mode—using general input parameters to give conservative, worst-case results—or they can be configured to provide more refined and realistic results. Refined modeling, however, requires a higher level of knowledge regarding input parameters, such as meteorology, topography of the area, and more details on the characteristics of the source (types of sprinkler nozzles, operating pressures, height above ground surface, etc.).

3.0.1 General Approach

The general approach adopted here was described in EPA guidance published in 1982 (EPA, 1982). The methods proposed by DEQ reflect improved treatment of some of the individual parameters.

The refinements proposed by DEQ include equipment-specific aerosolization factors; treatment of fine mist droplet transport, in addition to aerosolized pathogens; refined dispersion/deposition modeling; incorporation of risks due to ingestion of surface-deposited pathogens; and infectivity model estimates of the risk of infection. These refinements are described in greater detail in Section 6, starting on page 57.

3.0.2 Bio-Aerosol Prediction Model

The general expression for the concentration downwind of a microbial emission source such as a wastewater land application system (EPA, 1982) is:

$$C_d = D_d Q_s M_d + B$$

Equation 1

Where:

C_d = concentration of downwind location, in colony forming units per cubic meter of air (CFU/m³)

D_d = Dispersion factor, based on results of an atmospheric dispersion model (CFU/m³ per CFU/s)

Q_s = Aerosol source strength, adjusted for die-off of organisms (CFU/s)

M_d = Microorganism die-off factor

B = Background concentration (CFU/m³)

The terms Q_s , M_d , B , and D_d are discussed more fully in the following section(s).

An analogous equation is used to calculate surface deposition on downwind surfaces, such as produce. The general expression for the amount of microbes deposited on surfaces downwind of a microbial emission source, such as a wastewater land application system, is:

$$S_d = D_{dep} Q_s M_d + B$$

Equation 2

Where:

S_d = surface deposition flux of microbes at a downwind location, in colony forming units per square meter of surface (CFU/m²-hr)

D_{dep} = Deposition factor, based on results of an atmospheric dispersion/deposition model (CFU/m²-hr per CFU/s).

Q_s = Aerosol source strength, adjusted for die-off of organisms (CFU/s)

M_d = Microorganism die-off factor in air. Same as in Equation 1. In the risk calculations in Section 5, additional die-off on the plant surface is accounted for.

B = Background surface concentration of microbes (CFU/m²-s)

3.1 Aerosol Source Strength (Q_s)

The source strength, expressed in CFU emitted per unit time, is a function of the following:

- CFU content of the source material applied, in units of CFU per volume of material
- Application rate of source material, in units of volume per unit time
- Aerosolization efficiency, expressed as a fraction of source material

applied that forms aerosols of a size that can be advected downwind

- Initial organism die-off caused by the application mechanism

The following equation can be used to calculate the source strength:

$$Q_s = WFEI \quad \text{Equation 3}$$

Where:

- Q_s = aerosol source strength (CFU/s)
- W = CFU content of the source material (CFU/liter)
- F = Application rate (liter/s)
- E = Aerosolization efficiency (fractional)
- I = Impact factor (initial organism die-off, fractional)

The terms W, F, E and I, are discussed in the following section(s).

3.1.1 Pathogen Content of Source Material (W)

Downwind concentrations of pathogen CFUs vary directly with the CFU content of the source material. An accurate characterization of the CFU content and the variability of the CFU content for the specific material or materials applied is critical to characterizing the resulting risk.

In most instances, the CFU content of the source material applied will be based on sampling results—from either the source material itself or material from a similar operation that has been determined as representative. The source material for purposes of this document is wastewater. In general, samples should be collected, from locations in the process, during times that will yield results that are most representative of material generating aerosols. The following should be considered when developing a sampling plan to characterize the wastewater:

- *Sampling for appropriate organisms.* It is critical that the appropriate sample media and method be used to identify the specific CFUs, whether they are indicator organisms or specific pathogens. If indicator organisms are used, the relationship between concentrations of the indicator organism and the pathogen must be well established.
- *Sampling of all material types potentially used.* Sampling should be conducted from all source materials that may be land applied, in a manner that could generate aerosols. If there are multiple wastewater streams, then

samples should be collected from each stream.

- *Sampling intervals.* The wastewater should be sampled over a time interval such that variation in the wastewater is adequately characterized. For example, the CFU content of wastewater may vary considerably with season; therefore, sampling should be conducted during each season when it is land applied. If the wastewater will not be applied during the non-growing season, then sampling would not be necessary during this season.
- *Spatial variability.* Samples should be collected at sufficient locations to characterize any spatial heterogeneity of the wastewater.

3.1.2 Material Application Rate (F)

The material application rate is simply the rate at which material containing CFUs is applied or used by the process that generates aerosols. For land application by a sprinkler system, this would be the pumping rate to the sprinklers. Usually, this value can be measured directly. Variability in the application rate should be well characterized in the analyses. It is important to remember that the material application rate should correspond to only that portion of the irrigation configuration addressed in the dispersion modeling.

3.1.3 Aerosolization Efficiency Factor (E)

The aerosolization efficiency is the fraction of the material applied that results in aerosols that can be advected downwind. This parameter varies considerably with the type of application mechanism that generates aerosols and certain meteorological parameters, such as temperature, wind speed, and solar radiation.

Studies of impact type irrigation equipment at three locations indicated aerosolization efficiencies ranging from about 0.001 to 0.02, with median values of about 0.003 (EPA, 1982).

Data from the 1982 EPA study at one of the three sites were used to develop a relation for aerosolization efficiencies of rotating impact type sprinklers, based on the influence of wind speed, solar radiation and temperature. The relationship proposed by EPA in their 1982 guidance for rotating impact type sprinklers is:

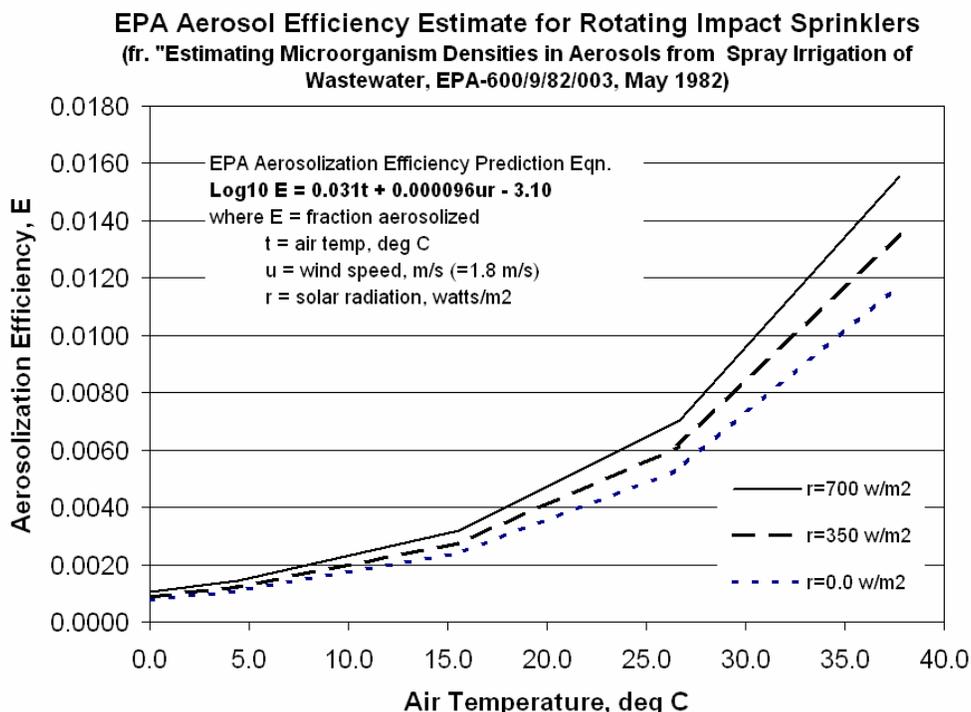
$$\text{Log}_{10}E = 0.031t + 0.000096ur - 3.10$$

Equation 4

Where:

- t = air temperature (°C)
- u = wind speed (m/s)
- r = solar radiation (watts/m²)

This relationship is depicted graphically in Figure 8 for a typical range of solar radiation and temperature values.



An alternative approach for estimating aerosolization efficiencies, discussed in Section 6, utilizes the equipment specific droplet size distributions of Kincaid (1989), along with his DRIFT02 temperature/ evaporation droplet drift model to determine which droplet size categories are aerosolizable under different conditions. This approach allows the modeler to explore different sprinkler types and pressures during design of the land application system.

3.1.4 Initial Organism Viability Reduction (or Impact) Factor (I)

Pumping and spraying will have an impact on the viability of microorganisms. High pressure and high impact systems may shock microorganisms as they exit

the nozzle and thereby decrease the viability. This process is different than the die-off factor M_d discussed in Section 3.2, which accounts for reduced viability because of environmental factors experienced as the aerosol is advected downwind.

Studies have indicated that microorganisms exhibit an initial rapid, second-order die-off during aerosolization, compared to the slower, first-order die-off that is represented by M_d . Values of the initial organism die-off, or impact factor (I) have been measured as ranging from 0.03 to 150 (EPA, 1982).

Estimates of the microorganism impact factor, duplicated from EPA (1982), are provided in Table 1. Measured values greater than 1.0 were likely a result of one of the following:

- Breakup of colonies through the aerosolization process (CFUs may be composed of multiple organisms – therefore, a single CFU may break into two separate CFUs)
- Toxic effects from constituents present in the land-applied material that inhibit growth when CFUs are present in that material, but are not present in aerosols in sufficient quantities to have a similar effect. Median values for fecal and total coliforms in EPA (1982) were 0.27 and 0.34, respectively.

Table 1. Estimates of microorganism impact factor, I (from EPA 1982)^a

<i>Microorganism</i>	No. Obs	Percentile Distribution of I Values						
		10%	25%	40%	50%	60%	75%	90%
Fecal Coliforms	13	NE ^b	0.14	NE	0.27	NE	1.2	NE
Total Coliforms	44	0.034	0.13	0.27	0.34	0.48	1.2	2.3
Standard Plate Count	33	0.076	0.23	0.40	0.44	0.50	0.74	2.5
Coliphage	43	0.036	0.20	0.38	0.71	1.1	1.9	3.8
Mycobacteria	8	NE	1.6	NE	1.9	NE	4.4	NE
<i>Clostridium perfringens</i>	11	NE	0.5	NE	2.5	NE	14	NE
Fecal Streptococci	31	0.57	1.5	2.0	3.6	5.7	13	67
<i>Pseudomonas</i>	13	NE	3.6	NE	29	NE	150	NE
Enteroviruses	2	NE	NE	NE	80 ^c	NE	NE	NE

Notes: a) All estimates based on data obtained at Pleasanton, Ca. b) NE = Insufficient number of samples to provide values at all percentiles, c) Approximate value.

3.2 Microorganism Die-Off Factor (M_d)

The die-off factor, M_d , accounts for the decay in microorganism viability as the aerosol is advected downwind from the application source. Factors affecting die-off are primarily temperature, relative humidity, solar radiation, and toxic compounds in ambient air. Conditions of low temperature, high humidity, and low solar radiation (night/rainy conditions) tend to favor microorganism survival. On the other hand, high temperature, low humidity, and high solar

radiation conditions tend to result in more rapid die-off, presumably due to desiccation, ultraviolet light and atmospheric oxidants. The die-off rate is also very organism specific.

The die-off factor can be calculated using the following equation (EPA, 1982):

$$M_d = e^{\lambda a_d} \quad \text{Equation 5}$$

Where:

M_d = microorganism die-off factor

λ = viability decay rate (s^{-1})

a_d = downwind distance (m)/wind speed (m/s)

Studies of the viability decay rate (λ) have shown ranges from -0.23 to about 0.0 . Median values for specific types of microorganisms ranged from -0.32 to about 0.0 , with values for fecal and total coliform at -0.023 and -0.32 , respectively.

Estimates of Viability Decay Rates from EPA (1982) are duplicated in Table 2. EPA (1982) suggested that the 40th percentile values may be more appropriate for daytime, dry, sunny conditions, while the 60th percentile values may be more appropriate for cooler, high humidity night or rainy conditions. The 50th percentile values could then be used for intermediate conditions.

The relative effect of EPA’s median decay rates on the microorganism decay factor, M_d , can be seen in Figure 9, where viability decreases at a faster rate with aerosol travel distance as the decay factor decreases.

Table 2. Estimates of viability decay rate, λ (from EPA 1982)^a

<i>Microorganism</i>	No. Obs	Percentile Distribution of I Values						
		10%	25%	40%	50%	60%	75%	90%
Total Coliforms	44	-0.23	-0.094	-0.050	-0.032	-0.02	-0.004	-- ^b
Fecal Coliforms	13	-0.19	-0.070	NE ^c	-0.023	NE	--	--
Coliphage	43	-0.11	-0.051	-0.029	-0.011	--	--	--
<i>Clostridium perfringens</i>	11	-0.10	-0.039	NE	-0.004 ^e	NE	--	--
Standard Plate Count ^d	33	-0.12	-0.020	-0.006	--0.004 ^e	--	--	--
Mycobacteria	8	-0.15	-0.009 ^e	--	--	--	--	--
<i>Pseudomonas</i>	13	-0.08	-0.008 ^e	--	--	--	--	--
Fecal Streptococci	31	-0.06	-0.006 ^e	--	--	--	--	--
Enteroviruses	0	--	--	--	--	--	--	--

Notes: a) All estimates based on data obtained at Pleasanton, Ca., b) -- = Slow decay rate, assume = 0.0 for model calculations, c) NE = Insufficient number of samples to provide values at all percentiles, d) Total aerobic and facultative bacteria, e) Questionable value, may be indistinguishable from zero.

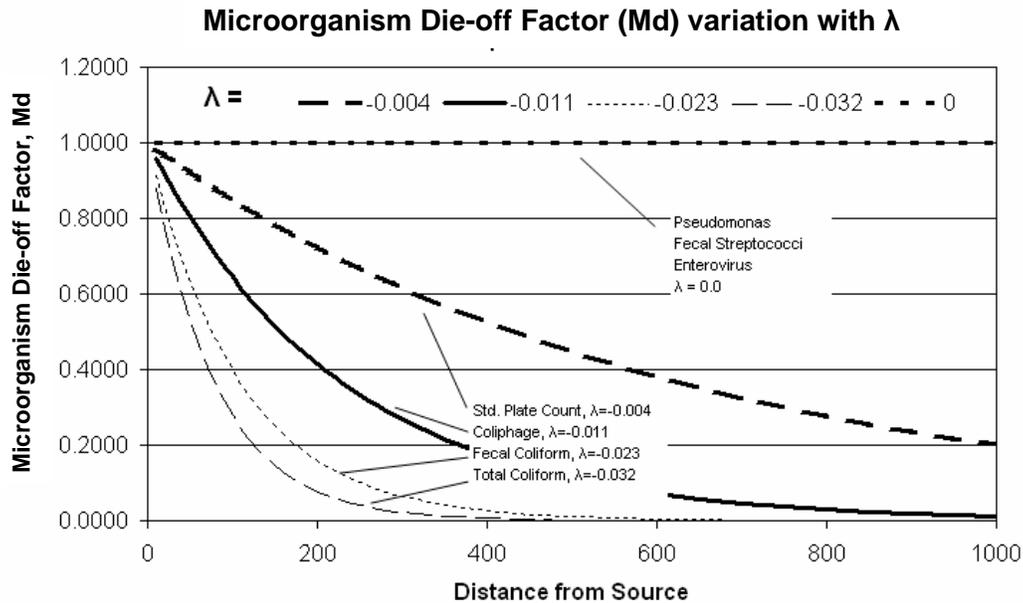


Figure 9. Microorganisms die-off factor, M_d , as a function of distance from the source for various decay rates (λ).

3.3 Microbial Background Concentration (B)

Background concentrations of most pathogens can be assumed to be negligible for many rural locations. If the background is uncertain, representative monitoring should be conducted to evaluate concentrations.

3.4 Atmospheric Dispersion Factor (D_d)

Aerosols are either transported downwind and deposited on surfaces or else taken up by an organism exposed to the material. As they are transported downwind, they are also subject to gravitational settling, diffusion, and larger-scale turbulence. These effects can be simulated by atmospheric dispersion models.

A dispersion model is a mathematical description of the meteorological transport and dispersion processes, using source and meteorological parameters, for a specific period in time. Calculations result in estimates of maximum constituent concentration for specific locations and times.

3.4.1 General Use of Dispersion/Deposition Modeling

Dispersion models have been commonly used to estimate downwind constituent concentrations caused by emissions from industrial smoke-stack type emissions. Through the modifications of model input parameters, a wide

range of constituent release types can be simulated by readily available dispersion models, including transport of aerosolized microorganisms.

3.4.2 Basis of Dispersion Models

The most commonly used type of models for regulatory purposes are the Gaussian steady-state models, which provide a steady-state solution to the transport and diffusion equations (transport [i.e. advection] + diffusion = dispersion). Steady state implies a basic assumption of constant emissions and constant meteorological conditions over the time step, and conditions of the previous time step do not influence results for subsequent time steps.

The basic Gaussian diffusion equation assumes:

- Atmospheric stability and all other meteorological parameters are uniform and constant throughout the layer into which the constituent is discharged, and, in particular, that wind speed and direction are uniform and constant in the domain;
- Turbulent diffusion is a random activity, and, therefore, the dilution of the constituent can be described in both horizontal and vertical directions by the Gaussian or normal distribution;
- The constituent is released at a height above the ground that is given by the physical release height and the rise of the plume due to its momentum or buoyancy (together forming the effective release height);
- Chemical mass reaching the ground level is reflected back into the atmosphere;
- The constituent is conservative, i.e., not undergoing any chemical reactions, transformation, or decay while in the atmosphere.

The spatial dynamics of pollution dispersion is described by the following type of equation in a Gaussian model:

$$C(x, y, z, t) = \frac{Q_m}{2\pi u \sigma_y \sigma_z} \exp\left(-\frac{y^2}{2\sigma_y^2}\right) \left[\exp\left(-\frac{(z - H_{eff})^2}{2\sigma_z^2}\right) \right] + \exp\left(-\frac{(z + H_{eff})^2}{2\sigma_z^2}\right)$$

Equation 6

Where:

- $C(x, y, z, t)$ = constituent concentration at point (x, y, z) at time step t;
 u = wind speed in the x, i.e., downwind direction, (m/s)

- σ = Standard deviation of the concentration in the y and z direction during time step t, i.e., in the cross-wind and vertical direction (m)
- Q_m = Emission rate (CFU/s)
- H_{eff} = Effective release height (after considering effects such as momentum induced plume rise and thermal buoyancy)

The steady state concentration at any point (x, y, z) in the modeling domain can be determined from the above equation using the constant emissions rate.

3.4.3 Dispersion Models Applicable for Modeling Aerosols in Typical Conditions

Either screening or more refined models can be used to estimate the impact of aerosol emissions to the atmosphere. Screening-level models require less site-specific input data and are easier to run, but they tend to give more conservative results, sometimes grossly over-predicting downwind concentrations by over an order of magnitude. Refined models require more data, including representative meteorological data, a detailed site plot plan clearly defining the facility property line and the location of the aerosol source, the location of potential exposed receptors, and elevation data for the facility and the surrounding area (if terrain effects will be considered).

A variety of models can be used to develop the dispersion factors (D_d) for use in the general bio-aerosol transport equation, as discussed in the following section. The factors themselves are generated by modeling a “unit emission rate” so that the concentrations predicted at each receptor are actually normalized, or based on a 1 CFU/s emission rate. By doing this, the concentration predictions at each downwind receptor $C(x,y,z,t)$ are transformed into the dispersion factors, D_d , which then possesses the units CFU/m^3 per CFU/s emission rate. Thus, the factor D_d in Equation 1 is a dispersion factor generated by the dispersion modeling result and the CFU emission rate (usually a unit emission rate) used in the modeling:

$$D_d = \frac{\text{Maximum Modeled Concentration (CFU/m}^3\text{)}}{\text{Modeled Emission Rate (CFU/sec)}} = \frac{C(x,y,z,t)}{Q_m} \quad \text{Equation 7}$$

Because there is a linear relationship between emission rates and downwind concentrations, it is not necessary to redo the modeling if emission rates change. Thus, the dispersion factor from the dispersion modeling output, based on a unit emission rate, can be used regardless of any change in emissions. The

dispersion factors then are specific to a given source size and configuration—and the atmospheric conditions of wind speed and stability class.

For refined models that also compute deposition, deposition factors should be generated and used in an analogous fashion to estimate microbial surface densities.

3.4.3.1 Box Model

The simplest and most conservative model for estimating concentration downwind from an area-type source is the box model, depicted in Figure 10. This model accounts for mixing of bioaerosols into a volume of air that begins at the downwind edge of the spray area or field, with width s , height, h , equal to the top of the spray envelope, and the length of the box is defined by the distance in meters that the wind travels in one second at wind speed, U . The box model equation is given by:

$$C = \frac{Q_s}{uhs} \quad \text{Equation 7}$$

Where:

- C = constituent concentration at all points downwind;
- u = wind speed along the “ l ” side of the box (m/s)
- h = actual height of the spray volume from the ground (m)
- s = length of the sprayed field, perpendicular to the wind (m)
- Q_s = microbial emission rate (CFU/s)

THE BOX MODEL:

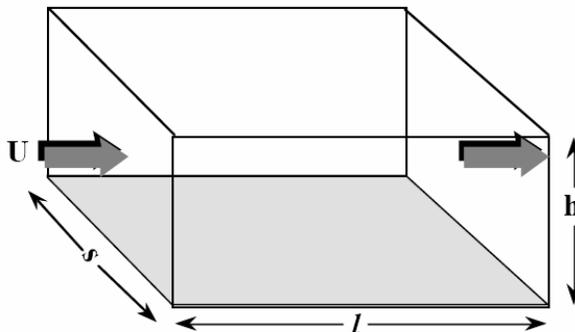


Figure 10. Box model for estimating downwind concentration.

The box model provides only a gross estimate; for a more precise estimate, other models can be used.

3.4.3.2 **Screening-Level Dispersion Model**

SCREEN3 is the most common screening-level model used for regulatory permitting purposes. It is available from the EPA at the following address:

<http://www.epa.gov/ttn/scram/>

The model runs in DOS mode and prompts the user for specific inputs.

This model can only simulate a single emission source and will only estimate plume-centerline concentrations at downwind distances. The user either specifies a wind speed and atmospheric stability class, or selects an option for the model to use worst-case conditions for the specified downwind distances. Model output is a 1-hour average concentration at one or more downwind locations.

If deposition is not important, this model may be used. For the example analysis provided in Section 7, a more refined method was used.

3.4.3.3 **Refined Dispersion Model**

The *Industrial Source Complex Short Term* model (ISCS3) and the *American Meteorological Society/Environmental Protection Agency Regulatory Model Improvement Committee* (AERMIC) *Dispersion Model*, AERMOD, are the primary refined models used for industrial point sources. These two models are very similar, with AERMOD using improved meteorological algorithms and terrain handling algorithms. AERMOD is proposed as the replacement model for ISCS3.

EPA approved dispersion models can be downloaded from the following address:

<http://www.epa.gov/ttn/scram/>

Because of the complexity of these models, many analysts purchase front-end graphical user interface (GUI) programs that simplify the running of these models and quality assurance measures.

Hourly meteorological data are used for both models. Unlike SCREEN3, these models calculate concentrations and surface deposition on a three-dimensional basis. Therefore, it is necessary to have an accurate map of the site and surrounding area to define emission source locations, property boundaries, and important receptor sites.

More details on model input data are provided in the following section(s).

3.4.4 Data Needs and Model Setup for Dispersion Modeling

Data needs for modeling heavily depend on whether screening-level or refined models are used. Screening models use conservative assumptions for many input parameters while refined models rely more on site-specific data.

3.4.4.1 Emission Rate

The dispersion models are designed to generally accept emission rates in terms of grams per second (g/s) to produce output in terms of micrograms per cubic meter ($\mu\text{g}/\text{m}^3$) for concentration and micrograms per square meter per second ($\mu\text{g}/\text{m}^2\text{-s}$) for surface deposition. To configure the model to provide output in terms that are equal to CFU/m^3 (for concentration) and $\text{CFU}/\text{m}^2\text{-s}$ (for deposition) the input rate must be in terms of 10^6 CFU per cubic meter (i.e. 1 CFU/s would be $0.000001 * 10^6$ CFU/s). When using refined models, a default factor of 10^6 is used in the model to convert output from g/m^3 to $\mu\text{g}/\text{m}^3$. To enter emissions in terms of CFU/s and obtain output in terms of CFU/m^3 , the default factor must be changed to 1.0.

3.4.4.2 Source Configuration

Land application systems are not typical sources modeled by the readily available dispersion models. These models have been designed primarily to simulate emissions from industrial smoke stacks. However, methods can be used to reasonably simulate emissions from various land application systems.

Aerosol emissions from sprinkler operations can be modeled as a series of volume sources, often called *virtual point sources*. A volume source is handled similarly to a point source in the model, except it is assumed the plume has dispersed to a specified size at the release point. In this manner, the plume behaves as if it were released from a point located further upwind of the release point (a virtual point).

The emissions source input parameters for a volume source are as follows:

- *Release Height* (m). The distance from ground level to the center of the volume defining the source should be used as the release height. This may be the sprinkler head location, depending on the characteristics of the sprinkler.
- *Initial Dispersion Coefficients*. These define the size of the volume source, establishing the standard deviation of concentration distribution in both the horizontal and vertical direction. For many sprinkler operations, the

volume source is most appropriately defined as a square box with the length of sides equal to the sprinkler application area dimensions. The initial dispersion coefficient in the horizontal direction (σ_{y0}) is calculated as the length of the side divided by 4.3 for a single volume source (for example, one end gun) or divided by 2.15 for a series of volume sources (for example, a row of sprinklers). The initial dispersion coefficient in the vertical dimension (σ_{z0}) is calculated as the height of the region of initial spray divided by 4.3 (EPA, 1995).

- *Source Location(s)*. Coordinates for actual and potential source locations must be defined for refined modeling. Screening-level modeling only requires that the minimum downwind distance to ambient air or the nearest potential exposure be defined, since screening-level modeling only calculates plume centerline concentrations.

The source must be identified by a horizontal coordinate system for refined modeling. Typically, *Universal Transverse Mercator* (UTM) coordinates are used. However, any coordinate system can be used, provided the same system is used for all sources, receptors, and identified site boundaries.

3.4.4.3 Site Layout and Receptor Locations

When using screening-level modeling, it is advised to select the option that directs the model to automatically determine the maximum downwind concentration between selected minimum and maximum receptor distances. Receptor distances should be bounded by the closest distance to ambient air or a potential exposure location and an additional downwind distance of at least 1,000 meters.

A scaled facility site layout will be necessary for refined modeling to correctly scale site boundaries, specific receptor locations, and the location of sources.

3.4.4.4 Terrain Data

Elevated terrain can have a substantial effect on concentrations observed at ground level. Both screening-level models and refined models can account for effects associated with dispersion over terrain features. When using SCREEN3, the user enters the terrain height above stack base (or in our application, sprinkler height) and the downwind distance on a receptor-by-receptor basis.

Terrain effects are included in refined modeling by specifying the terrain height of each receptor used in the modeling run. Many front end GUI programs for ISCST3 and AERMOD have capabilities of extracting elevations for receptors and sources from USGS Digital Elevation Model (DEM) files. Terrain effects are handled by a screening-level algorithm in ISCST3, and this may result in a substantial over-prediction of concentrations in some instances. Terrain handling algorithms in AERMOD are more refined; however, the data needs are more extensive with AERMOD, and the model is more difficult to set up and run than ISCST3.

3.4.4.5 Meteorological Data

Actual meteorological data are only required for refined models. SCREEN3 uses internally calculated worst-case meteorological data to generate maximum plume centerline concentrations for 1-hour averaged concentrations. Variability in wind direction is accounted for in longer averaging periods by using persistence factors as described in Section 3.4.4.6.

Hourly monitored meteorological data are used in refined models. These data are typically collected at *National Weather Service* (NWS) sites located at major airports. Preprocessor programs are then used to format the meteorological data for use in the models. Meteorological data sets and the preprocessor programs are available from the following:

<http://www.epa.gov/ttn/scram/>

The following are meteorological parameters are used in the model ISCST3:

- *Time* – year, month, day, hour.
- *Wind Speed* – Wind speed is typically used in units of meters per second. Gaussian models do not simulate periods of calm winds, and the models are constructed to exclude such periods during the calculation of concentrations for various averaging periods.
- *Wind Direction* – Wind direction is typically specified in data sets as the direction from which the wind is blowing. The models use wind vectors, with wind direction defined as the direction toward which the wind is blowing. The preprocessor programs make this adjustment automatically.
- *Temperature* – specified as absolute temperature, in degrees Kelvin.
- *Stability Class* – Atmospheric stability greatly affects how constituents

disperse through the atmosphere. Thermal turbulence can bring constituents from an elevated release down to ground level, and a highly stratified atmosphere can transport a plume miles downwind with relatively little dissipation. ISCST3 uses six stability classes to account for atmospheric stability affects on dispersion. Stability classes are set by the meteorological preprocessor program and consider wind speed, time of day, solar insolation, and cloud cover.

- *Mixing Height* – This specifies a cap on the extent of vertical dispersion, thereby accounting for thermal inversions. Mixing heights are calculated by the preprocessor program using twice daily upper air soundings recorded at major airports.

Many wastewater land application sites are not located near a major airport where National Weather Service meteorological data are collected. The challenge is then to find alternate data that are reasonably representative of the area. Wind direction and wind speed data are often available from small local airports near an application site. These data can be compared to potentially representative full data sets from National Weather Service sites to evaluate which data to use as model input. ISCST3 also has an option where the wind vectors can be rotated by a specific degree. This is especially useful for modeling within valleys.

3.4.4.6 Output Specifications

Model input for dispersion models is specified as grams per second (g/s) and output is typically expressed as micrograms per cubic meter ($\mu\text{g}/\text{m}^3$), as explained in Section 3.4.4.1. When modeling CFUs, the input would be either CFU/s or 10^6 CFU/s, and output would be in units of CFU/m^3 . The analyst must carefully check that the proper emission-to-concentration conversion factor is used. The default value is 10^6 , and this value cannot be changed for screening level modeling. Therefore, when using SCREEN3, the input must be in terms of 10^6 CFU/s to give output in CFU/m^3 . The factor can be changed from default for refined models such as ISCST3, thereby allowing input in terms of CFU/s and output in terms of CFU/m^3 .

Output for the screening level model SCREEN3 is in terms of a concentration averaged over a 1-hour period. Persistence factors can be used to estimate concentrations for longer averaging periods. These persistence factors have been developed by considering the amount of time conditions contributing to the 1-hour maximum concentration could persist during the alternate averaging

period. The following are persistence factors commonly used to convert maximum 1-hour concentrations to other averaging periods (EPA, October 1982):

1-hour to 3-hour factor	= 0.9
1-hour to 8-hour factor	= 0.7
1-hour to 24-hour factor	= 0.4
1-hour to quarterly	= 0.13
1-hour to annual	= 0.08

When using a refined model, maximum concentrations for other averaging periods are calculated directly from the hour-by-hour model output, using the actual hour-by-hour meteorological data to run the model.

FEBRUARY 2006

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4. Applicability of Existing Pathogen Dose-response Concepts to Wastewater

Dose-response assessment is a key component of quantitative risk assessment, providing a link between exposure to a hazardous agent, whether chemical or biological, and the probability of an adverse health effect.

4.0 Introduction

Microbial pathogen dose-response information comes primarily from ingestion studies on human volunteers. The infective dose estimate that comes from human volunteer studies is often presented as a minimum infective dose, but in fact it is usually the *median infective dose* (ID50), the dose at which infection occurs in 50 percent of the test animals or human volunteers exposed. It is not known what risk might be associated with lower doses that would likely occur in a land application situation. Another area of uncertainty is the potential risk to sensitive subpopulations, such as young children, the elderly, or immunocompromised individuals.

During land application of wastewater, any microorganisms in the wastewater are released into the environment. If pathogenic microorganisms are present, there is some potential for individuals to be exposed to these organisms. In such a situation, it is desirable to be able to estimate the risk of infection from exposure to pathogens. A dose-response model can be used to extrapolate risk from the higher doses used in infectivity experiments to lower doses that are more likely characteristic of environmental exposure, in a way similar to the approach used in chemical risk assessment. This allows risk to be estimated at low dose levels that would be impractical to test in empirical studies because of the need for unrealistic sample sizes.

However, dose-response analysis can be performed only if, following exposure, there is some kind of quantifiable response, such as infection or illness. For this reason, dose-response relationships can only be investigated for pathogenic organisms, and such relationships are microorganism-specific. Derivation of a dose-response relationship is not possible for an indicator species unless that species is itself pathogenic in addition to being an indicator of a specific kind or origin of contamination (e.g., fecal contamination). For

example, total coliform refers to a group of organisms, some of which may be pathogenic. Total coliform cannot have a dose-response relationship, but such a relationship can exist for individual enteropathogenic coliform species. For this reason, determining levels of total coliform in wastewater is not directly applicable to quantitative microbial risk assessment.

Wastewaters that are based on different processes are expected to contain different kinds of microorganisms and will likely vary in their potential to contain pathogens, as well as the kinds and concentrations of pathogens and temporal variation in their occurrence. It is not likely that quantitative dose-response information will be available for every kind of pathogen that might be found in every kind of wastewater.

Traditionally, total coliform or fecal coliform has been monitored at facilities to determine if a potential problem exists. These general classes of organisms are indicator species and, as a general class, are not pathogenic. Monitoring for a pathogen, however, that has the potential to occur in the wastewater, and for which dose-response information exists, such as pathogenic *E. coli*, can allow a risk estimate to be made. This results in a quantitative basis for adjusting parameters of wastewater land application.

It may not be possible to determine the total health risk associated with wastewater land application, because that would entail identification of every bacterial and viral pathogen in the wastewater and would require dose-response information for all of them. The monitoring plan would have to address all of these pathogens in order to estimate the probability of infection for each one, and the cumulative risk from exposure to all of them.

If only one or several pathogens are present, and if adequate dose-response information is available for them, then uncertainty associated with risk estimation is low. In the absence of such information, another way to address uncertainty would be to assume total coliform consists entirely of pathogenic coliform species; this is a very conservative approach, but it would ensure that risk is not underestimated.

4.1 Dose-response models vs. the Minimum Infective Dose Concept

Prior to the first application of dose-response models to the results from human feeding studies in the early 80s, the predominant theory among researchers was that there is an exposure threshold below which infection cannot occur.

This view is associated with use of the term *minimum infective dose*. More recently, the idea that a single pathogenic organism can cause infection has gained support (Haas 1983; Regli et al. 1991; Haas et al. 1993).

Infection is defined here as the process in which a microorganism multiplies within or on the host (Haas et al., 1999). Infection may or may not result in disease. The concept of a single organism being sufficient to cause infection is important with respect to risks associated with exposure to pathogens that might occur as a result of wastewater land application, because an individual may ingest or inhale a small number of viable organisms.

The concept of minimum infective dose might suggest one option for quantitative microbial risk assessment. It might be possible, for example, to estimate the likelihood that an individual receptor might receive a dose equal to or greater than the minimum infective dose. This would entail a process similar to that used in assessing risk associated with exposure to non-carcinogenic chemicals, in which the dose a receptor is estimated to receive is compared to a *reference dose*. A reference dose is essentially considered a safe dose. It is typically based on a dose in animal testing at which no adverse effects are observed. That dose might be reduced, through the application of uncertainty factors, to provide a margin of safety for use in human health risk assessment.

There are, however, several problems with applying this approach to microbial risk assessment. First, minimum infective dose usually refers to the ID50. It is not a no-effects level, therefore. Secondly, published minimum infective doses for pathogen species often vary considerably by source, and are often presented as ranges. See Table 3 for examples.

The solution to this problem is to use a mathematical model of infectivity. These models can be fit to experimental data, and they have the advantage of being able to predict risk of infection at doses lower than those that can be practically utilized in experiments. They also can take into account heterogeneity in the host-pathogen interaction. As is the case with chemical dose-response models, individual models cannot be proved or disproved, so determination of which model best describes the dose-response relationship in experimental data can involve some professional judgment.

See Section 4.2 for dose-response model evaluation and recommendations.

Table 3. Minimum infective doses for selected bacterial and protozoan pathogens.

<i>Pathogen</i>	<i>Minimum Infective Dose</i>
<i>Salmonella</i> spp.	As few as 15-20 cells; depends on age and health of host, and strain differences among members of the genus.
<i>Campylobacter jejuni</i>	400-500 cells in some individuals; depends on host susceptibility.
<i>Listeria monocytogenes</i>	Unknown; believed to vary with strain and individual susceptibility; may be fewer than 1,000 cells.
<i>Shigella</i> spp.	As few as 10 cells.
<i>Streptococcus</i> spp. (Group D: <i>S. faecalis</i> , <i>S. faecium</i> , <i>S. durans</i> , <i>S. avium</i> , <i>S. bovis</i>)	Greater than 10 ⁷ cells.
Enterotoxigenic <i>Escherichia coli</i> (ETEC)	100 million to 10 billion cells.
Enteropathogenic <i>E. coli</i> (EPEC)	Greater than one million cells.
Enterohemorrhagic <i>E. coli</i> O157:H7 (EHEC)	Unknown, but may be similar to <i>Shigella</i> spp. – as few as 10 cells.
Enteroinvasive <i>E. coli</i>	Thought to be as few as 10 cells.
<i>Cryptosporidium parvum</i>	Less than 10 organisms; presumably one organism can initiate infection.
<i>Giardia lamblia</i>	Ingestion of one or more cysts may cause disease.

Modified from: Foodborne Pathogenic Microorganisms and Natural Toxins Handbook. The “Bad Bug Book.” U.S. Food and Drug Administration, Center for Food Safety and Applied Nutrition.

<http://vm.cfsan.fda.gov/~mow/intro.html>

4.2 Recommended Dose-response Models

A population exposed to microorganisms in a given media at a given average concentration will receive a distribution of doses. Infection requires two sequential processes to occur (Haas 1983):

1. An individual ingests or inhales one or more organisms that are capable of causing disease.
2. Host-pathogen interaction results in the inhibition or death of some organisms, with some fraction of the dose surviving to initiate infection.

The probability of infection, if these two processes are regarded as independent, is the product of their individual probabilities (Haas et al. 1999). The probability of ingesting exactly j organisms from an exposure in which d is the mean number of organisms, is given as $P1(j | d)$, and the probability of k

organisms surviving from the initial j organisms to initiate infection is $P_2(k | j)$. The overall probability of k organisms surviving to initiate infection is then:

$$P(k) = \sum_{j=1}^{\infty} P_1(j|d)P_2(k|j) \quad \text{Equation 8}$$

Variability in individual doses is represented by the function P_1 , while variability in host-pathogen interaction is represented by P_2 . It is assumed that infection requires some minimum number of organisms, possibly as few as one. Denoting the minimum as k_{\min} , the probability of infection after being exposed to an average dose is:

$$P_i(d) = \sum_{k=k_{\min}}^{\infty} \sum_{j=k}^{\infty} P_1(j|d)P_2(k|j) \quad \text{Equation 9}$$

The minimum number of organisms surviving to initiate an infection, k_{\min} , may be large, or it may be as small as one. There are two hypotheses regarding the nature of infection initiation. The *hypothesis of independent action* holds that one organism can initiate infection ($k_{\min}= 1$). In the alternative hypothesis, called the *hypothesis of cooperative interaction*, k_{\min} is some number other than one. In this view, the combined effect of multiple organisms is necessary to initiate infection. Two models will be discussed here which are based on the assumption that one organism can initiate infection ($k_{\min}=1$). Each of these models has been found to fit dose-response data for a number of enteric bacterial and viral pathogens.

The distribution of organisms within the administered dosage in most experimental ingestion studies may be regarded as Poisson (Regli et. al 1991). If it is assumed that one organism is sufficient to cause an infection ($k_{\min} = 1$), and if host-microorganism interactions are constant, then the probability P_i resulting from ingestion of a single volume V of liquid containing an average of μ organisms per unit volume may be given by:

$$P_i = 1 - \exp(-r\mu V) \quad \text{Equation 10}$$

In this exponential model, r is the fraction of ingested microorganisms that survive to initiate infection. Most experimental dose-response data show a more gradual response to increasing dose than is predicted by Equation 10. The parameter r is a function of host-pathogen interaction, and is likely to be

affected by variations in host immune competence as well as variations in virulence among individual microorganisms. An assumption that r is characterized not as a discrete value but as a distribution of values, specifically a beta distribution, led to the development of an alternate dose-response model, the *beta-Poisson* model (Haas 1983):

$$P_i = 1 - \left(1 + \frac{\mu V}{\beta} \right)^{-\alpha} \quad \text{Equation 11}$$

The parameters α and β characterize the dose-response curve, and they are determined as best-fit parameters from human dose-response studies.

The beta-Poisson model has been shown to fit experimental data for a number of pathogens. For others, the exponential model provides a better fit.

Table 4 lists some microbial pathogens for which experimental data has been fit to the beta-Poisson or exponential model.

Table 4. Best-fit dose-response parameters from enteric pathogen ingestion studies.

Microorganism	Best model	Model parameters
Echovirus 12	Beta-Poisson	$\alpha = 0.374$; $\beta = 186.69$
Rotavirus	Beta-Poisson	$\alpha = 0.26$; $\beta = 0.42$
Poliovirus I	Exponential	$r = 0.009102$
Poliovirus I	Beta-Poisson	$\alpha = 0.1097$; $\beta = 1524$
Poliovirus III	Beta-Poisson	$\alpha = 0.409$; $\beta = 0.788$
<i>Cryptosporidium</i>	Exponential	$r = 0.004191$
<i>Giardia lamblia</i>	Exponential	$r = 0.02$
<i>Salmonella</i>	Exponential	$r = 0.00752$
<i>Escherichia coli</i>	Beta-Poisson	$\alpha = 0.1705$; $\beta = 1.61 \times 10^6$

Adopted from Gerba (2000), as modified from Regli et al. (1991).

The beta-Poisson model is ‘probably’ the most applicable for regulatory use for many pathogens because of its simplicity, though it has been noted that it is only an approximation of the actual single-hit dose-response relation (Teunis and Havelaar 2000), and that it can produce unrealistically high risk estimates at low doses. For regulatory purposes, this is not as serious a flaw as would be risk underestimation; however, in addressing uncertainty of a risk estimate based on this model, it should be remembered that the risk may be over-predicted at low dose levels.

Other models may be considered by DEQ if they are demonstrated to fit empirical data well, and if parameter values are available in the literature for pathogens of concern in Idaho wastewaters that are land-applied. There are a

FEBRUARY 2006

number of other possible models that utilize probability distributions other than the beta distribution to describe variation in the probability of survival and growth of individual pathogens within the body; see Haas et al. (1999) for a review.

FEBRUARY 2006

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5. Performing Microbial Risk Assessments

Building upon the existing concepts explored in the previous sections, specific recommendations can be developed for performing risk assessment as it applies to microbial pathogens.

5.0 Recommendations – An Introduction

Assessment of risk from microbial constituents in land applied wastewater has several points of similarity to the assessment of environmental exposure to chemicals. Such assessments generally follow a protocol (NAS 1983) consisting of the following steps:

1. Hazard Identification
2. Toxicity Assessment (Dose-Response)
3. Exposure Assessment
4. Risk Characterization

Each of these steps can be applied in quantitative microbial risk assessment (MIRA) as well. The steps will be discussed briefly below, followed by a more detailed description of the MIRA process, including the types of information required to conduct a risk assessment.

5.0.1 Hazard Identification

In chemical risk assessment, hazard identification involves cataloguing hazardous chemicals to which individuals (receptors) might be exposed, based on knowledge of processes that used or produced the chemical; historical site information, such as spills or other known releases to the environment; and monitoring data. The result is a list of *chemicals of potential concern* (COPCs). Similarly, in MIRA for land-applied wastewater, process knowledge and monitoring data can be used to identify microorganisms of potential concern.

5.0.2 Dose-Response

In chemical risk assessment, dose-response information generally comes from animal testing, as well as limited data from human studies, such as those evaluating occupationally exposed receptors, when such data are available.

Human data are preferred, but of necessity animal data are used most frequently. Because of this reliance on data from toxicity tests in animals, in which high doses are utilized in order to have an observable number of responses without the use of impractically large sample sizes, it has been necessary to develop models to extrapolate responses down to the lower doses that typically characterize environmental exposures. A number of models have been applied; currently EPA uses a linearized multistage procedure (EPA 1986, Crump et al. 1976) to develop cancer potency factors for carcinogenic chemicals.

Dose-response information for MIRA is based more often on studies using human volunteers than on animal test data. Because potential environmental exposures resulting from land application may be lower than the median infective dose from a volunteer study, it is necessary to apply a dose-response model in order to estimate the response at low-doses. As discussed in Section 4, appropriate models are based on the best fit to experimental data, and it is likely that different pathogenic microorganisms will require more than one infectivity model. (See Table 4, page 40, for representative models.)

A major difference between chemical and microbial pathogen dose-response testing has to do with temporal aspects of the dose-response relationship. In chemical toxicity tests, exposure to the chemical occurs either over a short portion of the test animal's lifespan (*acute exposure*), over a significant fraction of that lifespan (*chronic exposure*), or for some intermediate time period (*subchronic exposure*). Generally, environmental risk assessment utilizes chronic dose-response data, because of an assumption that human receptors will have exposure over a number of years. This chronic exposure is typically assumed to occur for 30 years. In pathogen infectivity testing, human volunteers receive a given dose, and infection either follows after a relatively short time period or it does not occur at all. Repeated exposures can be assumed to have independent probabilities of infection, although in actuality repeated exposures may result in a decreased probability of infection through increased *immune competence* (acquired immunity).

5.0.3 Exposure Assessment

Exposure to pathogenic microorganisms differs from exposure to chemicals in several ways. Chemicals can be persistent in the environment, or they can break down to yield other chemicals that may have greater or lower toxicity. Following exposure, which can occur through multiple routes such as

ingestion, inhalation, or dermal uptake, a chemical can be sequestered in the body, excreted, or metabolized to other molecules. These metabolites in turn may be sequestered or excreted, and they may be more or less toxic than the parent compound.

Pathogenic microorganisms released into the environment are subject to numerous stressors such as ultraviolet (UV) radiation, temperature, and humidity that are not favorable to their survival and growth; so generally they can be considered to have much lower persistence in the environment than chemicals. Pathogens can be taken into the body primarily by ingestion or inhalation, and unlike chemicals, microorganisms can reproduce in the body, and cause infection. Defense mechanisms of the host involve the immune system, rather than detoxifying enzyme systems such as hepatic microsomal mixed-function oxidase. A major difference in exposure modes results from the fact that an exposed individual who becomes infected can transmit the infection to other receptors. This kind of secondary transmission does not occur with chemicals, aside from the special case of a pregnant woman exposed to a chemical transmitting it to the fetus.

Despite differences between chemical and microbial exposure, the basic task of identifying environmental exposure pathways and exposure routes into the body is essentially similar. The exposure pathways and routes relevant to MIRA for land-applied wastewater will be discussed in detail later in this section.

5.0.4 Risk Characterization

Characterizing risk from chemicals involves estimating the dose to a receptor by means of exposure assessment, and then combining that estimate with dose-response information to develop an estimate of risk to the receptor, expressed typically as a probability in the case of carcinogenic chemicals, or a *hazard quotient* for non-carcinogens. In the latter case, the degree to which the estimated dose is greater or less than a “safe” dose is determined. In chemical risk assessment, exposure generally is assumed to be chronic, often occurring daily over a number of years.

In MIRA, characterizing risk requires identifying the endpoint of concern. Following exposure to a microbial pathogen, there is some probability of infection occurring. Infection can be defined here as survival and reproduction of the pathogen in the body. Some fraction of infected individuals may develop illness, and some fraction of those individuals may develop serious

complications, possibly leading to death. An important variable that affects the transition from infection to illness is the immuno-competency of the exposed individual. A decision is therefore required regarding whether the endpoint of concern is infection or overt illness.

The dose-response models described in Section 4 estimate the probability of infection. If protection of public health is interpreted to mean maintaining a low risk of illness from environmental exposures, an assumption can be made that simplifies the risk characterization process and still errs on the side of health protectiveness. This assumption is: if risk of infection is maintained at an acceptably low level, then risk of illness will likely be at a lower level, and therefore also acceptable. Given uncertainty about the likelihood of illness following infection, this should be a health-protective approach.

The MIRA process as presented here estimates the risk of an individual being infected from exposure to pathogenic microorganisms in wastewater aerosols. It does not consider secondary transmission, which refers to an individual being infected and then transferring the infection to another individual. This assumption is simplifying in that it avoids consideration of uncertain variables affecting the likelihood of secondary transmission (number and nature of contacts with other individuals, and whether an asymptomatic individual can transmit the infection). It is assumed that if the risk of primary infection from exposure to wastewater is maintained at an acceptably low level, secondary transmission will not increase risk to an unacceptable level. This assumption may not be entirely accurate, and it is recommended that this factor be considered in the further development of these risk assessment methodologies.

Microbial pathogen exposure resulting from land application can be considered as both a series of acute exposures, corresponding to land application events, and a more chronic component resulting from microorganisms surviving after land application following deposition on surfaces such as plants in vegetable gardens, or on other media that people might contact. In the case of pathogen-contaminated produce, the potential for infection has a longer duration than the land application event, but actual exposures are still acute in duration, consisting of meals containing the contaminated produce.

For the purposes of these recommendations, it is assumed that deposition of microorganisms on inanimate surfaces (*fomites*) with subsequent contact by hand followed by hand-to-mouth exposure is less significant than deposition on fruits and vegetables grown for human consumption. Therefore the exposure pathway involving fomites will not be assessed quantitatively. This

may result in some underestimation of risk. However, there would be considerable uncertainty associated with assessing this pathway quantitatively. One option would be to estimate concentrations on the soil surface from deposition, and then estimate a dose by assuming some rate of incidental soil ingestion. This would require an estimate of survival time on soil as a function of environmental stressors as well as the effect of other microbial species present in the soil at background levels.

Deposition on surfaces, such as children's toys, could also be estimated, as could the transfer efficiency from-toys-to-mouth or toy-to-hands-to-mouth. Lack of knowledge about these transfer efficiencies would result in high uncertainty. Skin infection from dermal exposure, although theoretically possible, is not assumed to be an important pathway; its quantitative assessment is also subject to high uncertainty. Given this uncertainty, and the expectation that the contribution to risk from these pathways would be less than that from direct inhalation or ingestion of airborne pathogens, or deposition of pathogens on food, it is appropriate to address these pathways qualitatively. It is anticipated that this would be required only in situations in which the major pathways indicate significant risk.

The probability of infection from wastewater land application, then, is a function of the probability of infection from inhaling or ingesting pathogens during an individual event, as well as the number of application events over time. Additionally, it is a function of deposition rates on homegrown produce in the yard of a residential receptor, survival of pathogens on surfaces of fruits and vegetables, and frequency of produce consumption. Risk can be calculated for each land application event, and annual and lifetime risks can then be calculated by combining risks from the two pathways over the time period of concern.

Another important part of risk characterization is an analysis of uncertainty. Risk estimates always have associated uncertainty, as discussed above, and this uncertainty can be discussed qualitatively, or analyzed quantitatively by means of *Monte Carlo* techniques.

The following sections will provide more detailed recommendations on conducting land-application MIRA.

5.1 Estimating Infectivity (Dose-Response)

Microbial pathogens potentially present in wastewater may present a risk of infection from ingestion, inhalation, or both exposure routes. The focus here is primarily on ingestion of enteric pathogens, but for some wastewaters the primary concern may be a respiratory pathogen. Characterization of the wastewater in terms of pathogen species will establish which pathogen or group of pathogens has the potential to present significant risk of infection. Dose-response models can then be selected for the pathogen(s) of concern.

- i) In Section 4, models were presented for the estimation of low-dose infectivity. Either the exponential or beta-Poisson model may be appropriate for a number of microbial pathogens (see Table 4, page 40). For others, different models may be selected from the literature as appropriate. Different models may be required for respiratory pathogens than those used for enteric pathogens. It will be necessary for a MIRA practitioner to have some familiarity with the literature in order to select models that are well-supported by data from experiments or analyses of disease outbreaks.

5.2 Estimating Exposure

Most of the literature in MIRA has dealt with ingestion exposure. While inhalation is clearly an important exposure route for a receptor exposed to a wastewater aerosol containing pathogens, it is less obvious whether ingestion exposure may also occur in this type of exposure scenario. This is an important question if the wastewater contains a pathogen that is known to be infective following ingestion, but either is thought not to be infective through inhalation, or its infectivity via inhalation is unknown.

Ingestion of enteric pathogens can occur following deposition of wastewater aerosols on produce, which is subsequently consumed by humans.

Additionally, a fraction of inhaled aerosol can actually contribute to the ingestion dose, based on the size of the inhaled particles. Particle deposition occurs in different regions of the respiratory tract by different mechanisms according to size (Amdur et al. 1991). Large particles (5 – 30 μm) deposit in the nasopharyngeal region by inertial impaction. Particles in this size range may consist of organic matter with attached microorganisms, or clumps of microorganisms. Deposition of particles from 2-5 μm occurs by sedimentation in the trachea, bronchi and bronchioles. Many individual bacterial cells are in

this range; *E. coli*, for instance, averages about 2 μm in length. Particles with diameters of 2 μm or less are deposited in the alveolar region, by diffusion.

Particles that deposit on mucus-coated nasopharyngeal surfaces may be swallowed. Larger particles that deposit in the tracheal, bronchial or bronchiolar regions may also be swallowed, through action of the mucociliary elevator. The epithelium lining of these regions contains ciliated cells. The mucus layer covering these cells is moved upward by beating of the cilia, to the mouth, where it is swallowed. The mucus contains deposited microorganisms, as well as macrophages which have engulfed microorganisms. Through this mechanism, inhaled enteric pathogens not destroyed by macrophages can gain access to the digestive tract. Mucociliary clearance of deposited particles is completed within 24 to 48 hours.

Ingestion dose of enteric pathogens as a fraction of inhaled dose can be estimated based on particle size distribution in the aerosol, with the assumption that particles in a certain size range will ultimately be ingested, as described above. Ingestion dose from aerosol pathogens can then be estimated by:

$$N_a = C_d \times IR \times F_{ing} \times ED_l \quad \text{Equation 12}$$

Where:

- N_a = Dose per land application event (CFU)
- C_d = Concentration of microorganisms in air (CFU/ m^3) from Equation 1
- IR = Hourly inhalation rate (m^3/hr)
- F_{ing} = Fraction of inhaled particles ingested
- ED_l = Duration of land application event (hr)

The concentration of microorganisms in air (C_a) is estimated through modeling (see Section 4). The default hourly inhalation rate (IR) is 1.6 m^3/hr , an appropriate value for adults engaged in moderate activity (EPA 1997). The recommended default value for F_{ing} of 0.8 is a conservative estimate based on the size distribution of aerosol particles as a function of wastewater solids content. In most wastewater aerosols, the particle size will be such that 80 percent of the particles are assumed non-respirable, but instead become part of the ingested dose. Land application *event duration* (ED_l) is a site-specific value.

Ingestion dose from consumption of homegrown produce can be estimated through modification of methodology in EPA (1998) for estimating aboveground produce concentration of chemical contaminants resulting from

deposition onto plant surfaces of particulates from hazardous waste combustion facilities. The following equation yields a concentration of pathogen colony forming units (CFU) on produce:

$$C_p = \frac{S_d \times R_p \times [1 - \exp(-k_p \times T_p)] \times ED_l \times EF_l}{Y_p \times k_p} \quad \text{Equation 13}$$

Where:

- C_p = Plant concentration expressed as dry weight (DW) due to deposition (CFU/kg DW)
- S_d = Microbial surface deposition flux (CFU/m²-hr)
- R_p = Interception fraction of edible portion of plant (unitless)
- k_p = Plant surface physical loss coefficient (yr⁻¹)
- T_p = Length of plant exposure to deposition per harvest of edible portion of *i*th plant group (yr)
- ED_l = Land application event duration (hr)
- EF_l = Land application event frequency (yr⁻¹)
- Y_p = Yield of standing crop biomass of the edible portion of the plant (kg DW/m²)

Plant concentration resulting from deposition, C_p , is estimated through modeling. The interception fraction of the edible portion of the plant (R_p) is related to plant productivity (Y_p) as well as the relative ingestion of different classes of produce. EPA (1998) derived a default weighted average value for R_p by using separate R_p values for exposed fruits and exposed vegetables, developed by Baes, Sharp, Sjoreen, and Shor (1984). The class-specific R_p values were then weighted by the relative ingestion of each class based on the EPA 1997 *Exposure Factors Handbook* (EPA 1997) to yield a weighted average R_p of 0.39.

Several physical processes can reduce the amount of bacteria that has deposited on plant surfaces from wastewater aerosol: wind removal, water removal, and growth dilution (EPA 1998). The term k_p is a measure of the number of bacteria lost from plant surfaces by these physical processes, and is given by:

$$k_p = \left(\frac{\ln 2}{t_{1/2}} \right) \times 365 \quad \text{Equation 14}$$

Where:

k_p = Physical loss coefficient (yr^{-1})

$t_{1/2}$ = Half-life (days)

365 = Units conversion factor (days/yr)

Based on a half-life based on physical processes only, EPA (1998) recommended a default k_p of 18 yr^{-1} .

It is likely that microorganisms are effectively removed from plant surfaces through death as a result of environmental stresses, such as ultraviolet radiation, and are thus subject to population decline from factors other than physical removal by wind and rain. It is appropriate, therefore, to include a biological loss coefficient in Equation 14. A complete review of currently available data on biological half-lives is beyond the scope of this document, but is necessary in order to select a default biological loss coefficient that would be appropriate for multiple pathogens, or pathogen-specific values.

EPA (1998) recommended a value of 0.164 yr for T_p , the length of time that produce would be exposed to deposition prior to harvest. This number is based on the growing season for hay rather than homegrown produce, and it represents a source of uncertainty. If information on time-to-harvest is available for specific produce of concern, a different value can be used for this parameter.

The duration (ED_i) and frequency (EF_i) of land application events are site-specific parameters.

Productivity (Y_p) is defined by Baes, Sharp, Sjoreen and Shor (1984) as:

$$Y_p = \frac{Y_{hi}}{A_{hi}} \quad \text{Equation 15}$$

Where:

Y_{hi} = Harvest yield of the i th crop (kg DW)

A_{hi} = Area planted to the i th crop (m^2)

EPA (1998) recommended a value of 2.24 kg DW/m^2 based on an ingestion rate-weighting of Y_p values for different classes of produce. The primary uncertainty associated with this variable is that site-specific values of Y_{hi} and A_{hi} may be different than those used to derive the default Y_p value.

Once the concentration of microbial pathogens on produce is determined, the dose from consumption of produce is given by:

$$N_p = C_p \times CF \times IR_p \times F_h$$

Equation 16

Where:

N_p = Daily dose from produce (CFU/day)

C_p = Concentration in produce (CFU/kg DW)

CF = Conversion factor (1kg/1,000g)

IR_p = Daily ingestion rate of produce (g/day DW)

F_h = Fraction of consumed produce which is homegrown (unitless)

The default value for IR_p is 30 g/day DW, based on the EPA 1997 Exposure Factors Handbook. (EPA 1997). Subsistence farmers can be assumed to grow all of their own produce ($F_h = 1$). For the general public, it can be assumed that 25% of consumed produce is homegrown (EPA, 1998).

5.3 Risk Characterization

Probability of infection can be calculated once doses are estimated for the microorganisms and exposure routes of concern; in this case, ingestion of inhaled aerosol particles and ingestion of homegrown produce containing bacteria from air deposition. One of the dose-response models described previously, the beta-Poisson model, is appropriate for a number of pathogens, and has the general form:

$$P_i = 1 - \left(1 + \frac{N}{\beta} \right)^{-\alpha}$$

Equation 17

Where:

P_i = Probability (risk) of infection

α , β = Parameters characterizing the host-pathogen interaction
and N is either:

N_a = Dose of pathogen from aerosol per land application event (CFU/event)

or:

N_p = Daily dose of pathogen from produce consumption (CFU/day)

The parameters α and β are taken from the literature, and are pathogen-specific. For other pathogens, such as *Salmonella*, P_i is determined using an exponential model:

$$P_i = 1 - \exp(-rN)$$

Equation 18

The parameter r , the fraction of ingested organisms that survive to initiate infections, is pathogen-specific and taken from the literature.

Once the probability of infection has been calculated for each pathway, the cumulative probability of infection per day (P_{ap}), assuming no more than one land application event occurs daily, can be calculated:

$$P_{ap} = P_a + P_p - (P_a P_p) \quad \text{Equation 19}$$

In this equation, P_a and P_p are the risks of infection from airborne and homegrown produce-containing microbial pathogens, respectively. For risk at levels typically considered acceptable in a regulatory context (typically from 1×10^{-6} to 1×10^{-4}), the third term, $P_a P_p$, becomes insignificant and Equation 14 effectively reduces to:

$$P_{ap} = P_a + P_p \quad \text{Equation 20}$$

In situations where there are multiple pathogens of concern, cumulative infection risk can also be calculated. In this case, the appropriate model and model parameters for each pathogen will be used to calculate risk per land application event and daily risk from produce consumption. It is possible that infection risk associated with individual pathogens might be acceptable, while cumulative risk from several pathogens is unacceptable. If cumulative risk is unacceptable, the separate risk estimates for each pathogen species may provide useful information for risk management.

The annual risk of infection can also be estimated for each pathway. It is probably more appropriate to base regulatory decisions on annual risk rather than risk per exposure event. There is precedent for the use of annual risk for regulatory decision-making. The EPA developed the *Surface Treatment Rule* in 1991, which established the goal of treatment to be a risk of *Giardia* infection not greater than 1 per 10,000 (10^{-4}) exposed persons annually.

Risk must be annualized separately for each pathway, as it is unlikely that the number of land application events per year will equal the number of days of produce consumption.

For ingestion of airborne pathogens, annual risk is given by:

$$P_{Aa} = 1 - (1 - P_a)^t \quad \text{Equation 21}$$

Where:

P_{Aa} = Annual risk of infection from airborne pathogens

P_a = Risk of infection per land application event

l = Frequency of land application events per year (site-specific)

Annual risk from homegrown produce ingestion is estimated similarly:

$$P_{Ap} = 1 - (1 - P_p)^n \quad \text{Equation 22}$$

Where:

P_{Ap} = Annual risk of infection from produce

P_p = Risk of infection from produce per day

n = Number of days/year with produce consumption

Produce ingestion frequency (i) is a function of growing season length and the types and quantities of fruits and vegetables grown. It is assumed that canning produce kills microbial pathogens; however, drying or freezing produce may not kill the organisms. It is assumed that produce consumption begins at some time after the start of the growing season, and extends for some time after the end of the season. A value of 120 days/yr is recommended as a reasonably conservative default value for this parameter. It may be appropriate to use a different value based on climatic conditions in different regions of the state.

Cumulative annual risk for both pathways would then be:

$$P_{Aap} = P_{Aa} + P_{Ap} - (P_{Aa} P_{Ap}) \quad \text{Equation 23}$$

In risk assessment of exposure to chemical carcinogens, lifetime risk is estimated. Lifetime risk can be estimated in MIRA, as well. In this case, the variables l and n in Equations 21 and 22 would be multiplied by 70 years as a default lifetime to calculate P_{La} and P_{Lp} . Equation 23 would then be used to calculate P_{Lap} , the lifetime cumulative risk of infection.

5.4 Uncertainty Analysis

Exposure estimates also carry significant uncertainty. Pathogen dose to receptors is based on, in the case of inhaled dose, such variables as source strength, dispersion, microbial die-off, as well as local background of the organisms in air. Each of these variables has associated uncertainty. Dose from produce ingestion is a function of the deposition rate onto plant surfaces, as

well as survival time on those surfaces. The assumption of a physical loss parameter only, and not a biological half-life on plant surfaces as well, probably results in some overestimation of risk from produce ingestion.

A few other examples of exposure parameters with significant associated uncertainty include the ingestion fraction of inhaled cells, the number of produce meals per year, and whether produce will be washed before consumption.

As in chemical risk assessment, a large part of the uncertainty of a risk estimate is associated with the dose-response relationship. If a model does not accurately describe the dose-response relationship for a species or strain of pathogen present in wastewater aerosol, risk may be over- or underestimated. Obviously if pathogens are present which have not been identified, or for which an infectivity model has not been developed, then risk will be underestimated. Some uncertainty derives from inherent properties of the models themselves. As discussed previously, the beta-Poisson model may over-predict risk in the low dose range.

Escherichia coli provides an example of the wide variability in infective potential of different strains. The reference for the beta-Poisson parameters cited in Table 4 (page 40) does not list relative percentages of different *E. coli* strains in the experimental doses. Enterotoxigenic *E. coli* may have infective doses in the range of hundreds of millions of cells. The infective dose of enterohemorrhagic *E. coli* O157:H7 may be only a few cells. Presumably *E. coli* (O157:H7) is relatively rare in most wastewaters, but this is a source of uncertainty as it is unlikely that wastewaters will be completely characterized by strain. Similarly, more than 2,000 serotypes of *Salmonella* have been identified. It is unlikely that a single model will describe the dose-response relationship equally well for all of these serotypes.

Chemical risk assessment deals with dose-response uncertainty through the use of uncertainty factors. For example, in developing reference doses for noncarcinogenic chemicals, a series of uncertainty factors of 10 is typically applied to animal test data, to account for interspecies extrapolation, protection of subpopulations which might have unusual sensitivity, and other factors, so that the reference dose may be up to 10,000 times lower than the highest no-effect dose in the animal study on which the reference dose is based. A similar method could be used to adjust the risk associated with a given dose of pathogenic microorganisms. Alternatively, *E. coli* identified wastewater, for example, could be assumed to consist entirely of O157:H7 rather than a

mixture of different strains containing a small percentage of O157:H7. A resulting risk estimate that precludes any land application of wastewater, and is therefore unacceptably high, could be a trigger to sample wastewater for this strain.

It is incumbent upon the MIRA practitioner charged with protection of public health not to underestimate health risk. Overall, it is likely that the parameters of the exposure assessment are conservative, in that they are unlikely to underestimate dose. The response associated with that dose carries greater uncertainty. If there is a concern that the risk assessment might underestimate risk it may be appropriate, on a case-by-case basis, to apply uncertainty or safety factors to the response expected with a given dose.

6. Implementation of a Combined Fate, Transport and Risk Analysis Tool

The scientific and technical background for conducting microbial risk analyses have been summarized in Sections 3, 4 and 5 of this report. Together, these elements define a process (Figure 11) for which it is possible to estimate microbial fate and transport, along with the potential daily and annual risks of infection due to inhalation and ingestion of pathogens originating from wastewater land application. DEQ has implemented equations found in the earlier sections in a spreadsheet format to provide reproducible microbial fate/transport and microbial risk estimates.

6.0 Limitations of the Tool

This application was developed as an exploratory tool to provide preliminary internal analyses of microbial levels and their potential human health risks. It should be used to explore risk variation as applied to actual wastewater application projects, and, perhaps, to supplement existing methods. However, it should not be used for stand-alone decision-making until considerable use, testing, and peer review has taken place.

This application is currently in a complex format suitable only for expert personnel having refined atmospheric modeling capabilities. However, this application may be used to explore the factors that contribute to microbial risk and to address the potential uncertainties that remain in the areas of microbial viability, aerosolized particle size and transport, surface deposition pathways, and both inhalation and ingestion risk estimates. It may also be used to prepare more condensed but user-friendly graphical tools for use by a wider audience to make microbial transport and perhaps risk estimates.

DEQ Microbial Risk Assessment (MIRA)

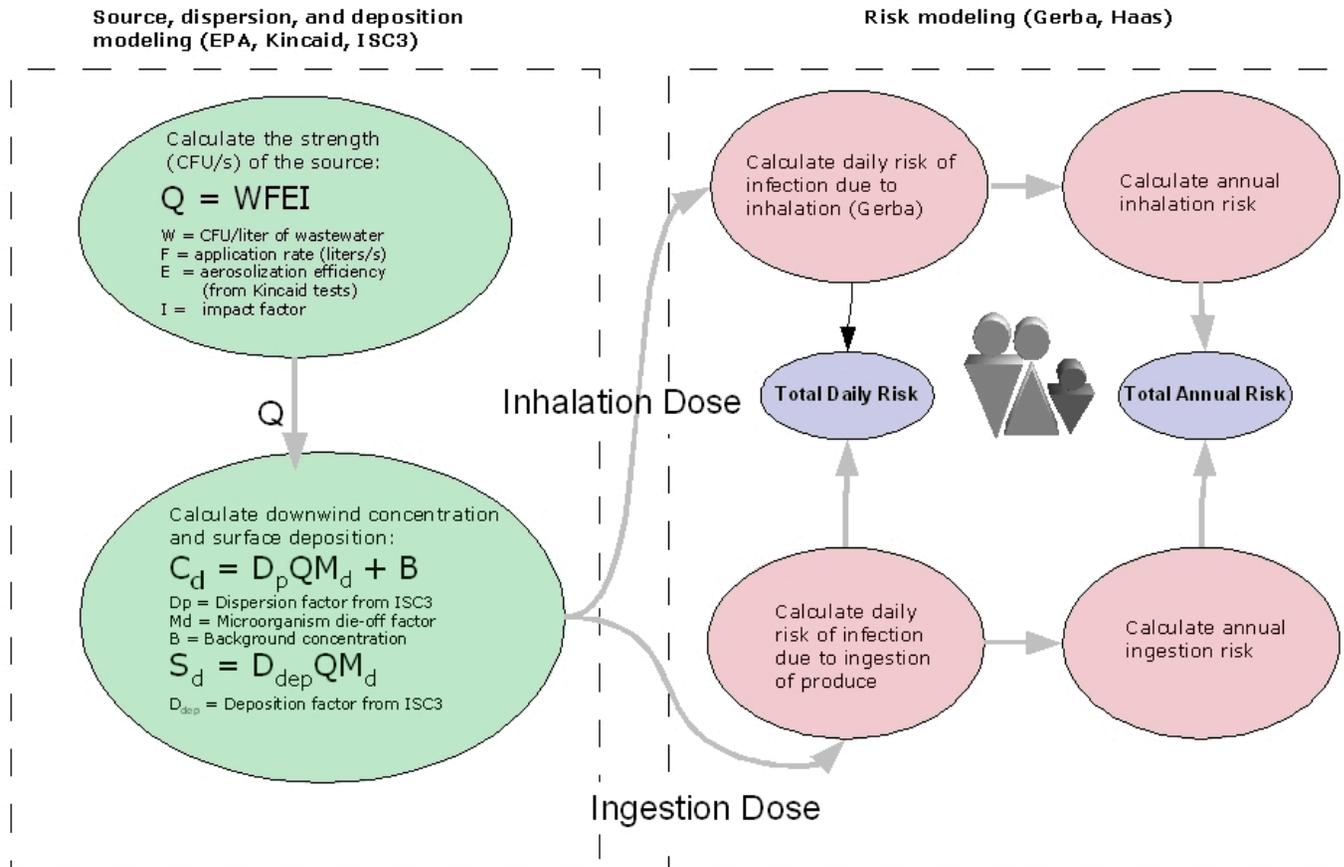


Figure 11. DEQ microbial risk assessment process.

6.1 Improvements Over the Original EPA Model

In view of the large uncertainties, the model is designed to be conservative in each area, however, refinements are certainly appropriate in many areas to improve the accuracy of the model.

There are five primary areas of improvement in this application over the original EPA (1982) model:

- Sprinkler specific aerosolization and fine droplet fractions
- Consideration of both mist droplets and aerosolized particle transport
- Refined dispersion modeling with the Industrial Source Complex Model (ISC3)
- Inclusion of a surface deposition pathway
- Human health risk estimates

These improvements are described in the following section(s).

Some of the improvements identified in this section may prove to be unnecessary or inappropriate, however they are included in this exploratory model so that DEQ may evaluate their importance and their utility in evaluating wastewater designs.

6.2 Aerosolization and Fine Droplet Fractions

Median aerosolization efficiencies were found to be 0.003, as discussed in Section 3.1.3, page 20 (EPA 1982). A relationship was also proposed by EPA to incorporate the influence of temperature, solar radiation and wind speed into aerosolization efficiency estimates.

This application uses a more refined approach to determine the fine droplet and aerosolized fractions of specific wastewater spray systems, using the spray droplet data and DRIFT02 model of Kincaid (1989).

6.2.1 Equipment Specific Droplet Size Distributions

An alternative approach to estimating aerosolization efficiency is to determine the fraction of very fine spray ($< 100 \mu\text{m}$) and assume that under most daytime conditions, droplets of this size will become 100 percent aerosolized. Dr.

Dennis Kincaid, U.S. Department of Agriculture Agricultural Research Service (USDA-ARS) in Kimberly, Idaho, has developed droplet size distributions for a wide range of sprinkler types, nozzle sizes and operating pressures (Kincaid, 1996).

Example size distributions are shown in Figure 12³ for a Nelson Model 85 Big Gun⁴, a Rainbird Model 30 rotating impact sprinkler, and a Senninger Wobbler⁵ nozzle.

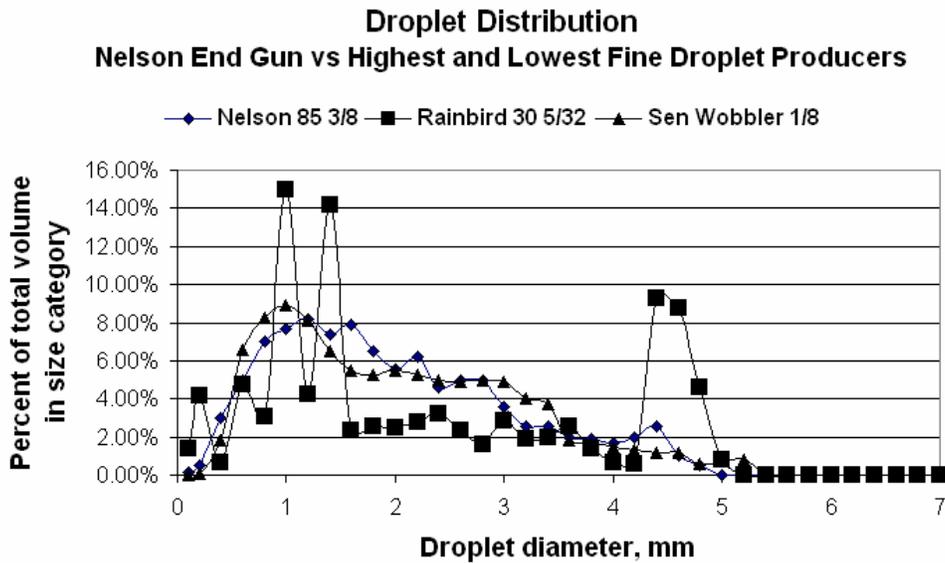


Figure 12. Droplet distribution.

Additional size distribution data in the smallest two size categories, 0.1 mm and 0.2 mm (or 100 μm and 200 μm), are depicted in Figure 13 for selected sprinklers that produce minimal fine droplets (0.1 mm and 0.2 mm). More typical endgun, rotator, and impact sprinklers are shown in the right hand side for comparison. Kincaid test identification numbers are shown across the bottom of the chart, along with the sprinkler designation.

³ The example sprinklers are provided only for the purpose of comparison. DEQ does not endorse any specific brand or model of sprinkler.

⁴ Big Gun is a registered trademark of Nelson Irrigation, Corporation.

⁵ Wobbler is a registered trademark of Senninger Irrigation, Corporation.

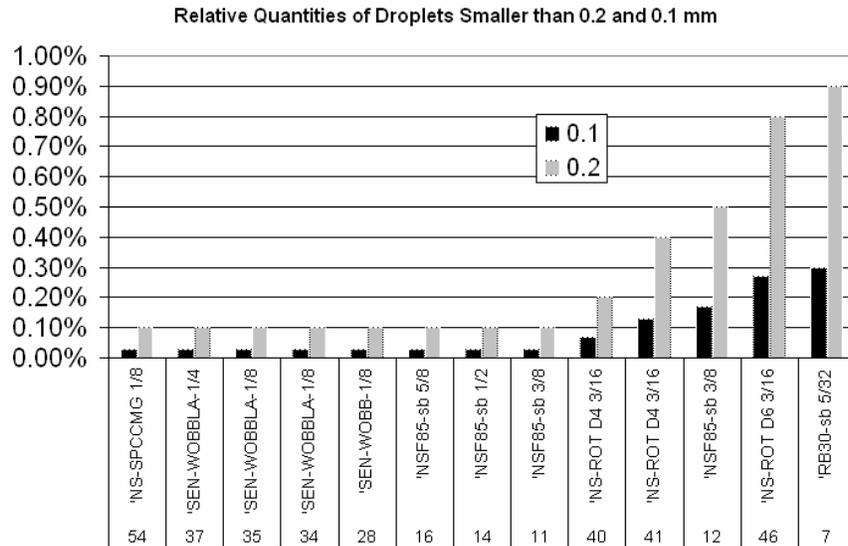


Figure 13. Relative quantities of droplets smaller than 0.2 and 0.1 mm for various sprinkler types.

6.2.2 DRIFT02 Temperature Evaporation Model

Kincaid has also developed a droplet evaporation and temperature model (Kincaid et al., 1989), which uses the measured droplet size distributions in a heat transfer/diffusion model to predict evaporation loss rates for each droplet size category.

The model version used here, DRIFT02, uses ballistic trajectories based on wind speed, nozzle velocity, and angle, to determine droplet trajectory, fall time, distance traveled, and droplet temperature and evaporation rate. For each droplet size category, DRIFT02 predicts the travel distance and evaporation loss before the droplet hits the ground.

Unfortunately, since this is a droplet model (developed for irrigation design purposes), it neglects aerosolized particles (after evaporation of water) and it neglects the effects of turbulent diffusion on droplet fall rates and removal rates. Thus, it is not suitable to use for predicting turbulent dispersion of fine droplets and aerosols.

On the other hand, DRIFT02 is suited for modeling the fate of the fine droplets generated by various sprinkler/nozzle configurations under various environmental conditions of temperature, humidity and wind speed.

The effects of atmospheric temperature and pressure on the percent of water loss for small droplets was explored by running the DRIFT02 model at various temperature and humidity conditions.

First, DRIFT02 runs were made to determine what size of droplets can evaporate to dryness under the lowest humidity levels that occur in most areas (10%). The results, shown in Figure 14, indicate that regardless of temperature, all droplets larger than 0.4 mm exhibit minimal water loss while 0.1 mm droplets evaporate to dryness at all temperatures (100% water loss). The 0.2 mm droplets show increasing significant water loss as the temperature approaches 100 °F, but they still never evaporate totally even at very warm and extremely dry conditions (100° F, 10% RH). Thus, it is clear that all droplets 0.2 mm and larger do not typically become aerosolized.

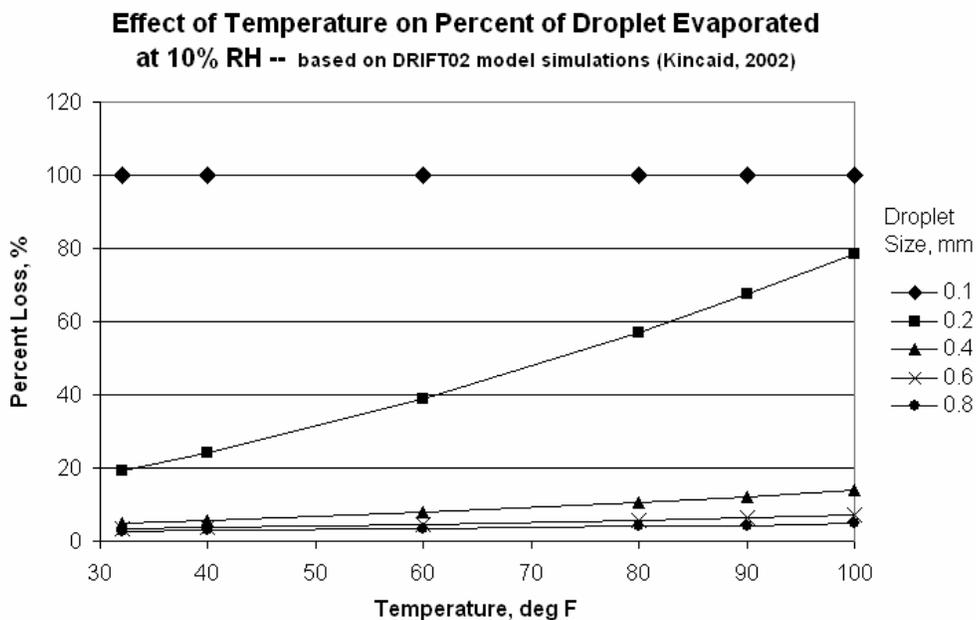


Figure 14. Effect of temperature on percent droplet evaporated.

Additional DRIFT02 runs were conducted at higher humidity levels to further evaluate the behavior of the smallest droplets, 0.1 mm and 0.2 mm under more typical ranges of relative humidity for the Idaho growing season. These results are shown in Figure 15.

Again, 0.2 mm droplets never evaporate, however, 0.1 mm droplets reach 100% water loss under a wide variety of conditions. During the growing season, when relative humidity levels are typically 20 – 60% and when temperatures are typically 60 – 100° F, the 0.1 mm droplets always evaporate to dryness. However, at night, or during rain storms, when humidity levels are typically above about 80% RH and temperatures are typically less than 80° F, the droplets do not evaporate to dryness and are thus not aerosolized.

This analysis suggests that 0.1 mm droplets are typically aerosolized during the daytime and not aerosolized during the night or during daytime conditions near the dew point (close to 100% RH). For computational ease, the DEQ model currently utilizes either the 0.1 mm droplet fraction or the same (0.1 mm) fraction as aerosolized particle dispersion (i.e. assigned a 20 μm diameter rather than 100 μm), whichever gives the greatest microbial concentrations at each distance.

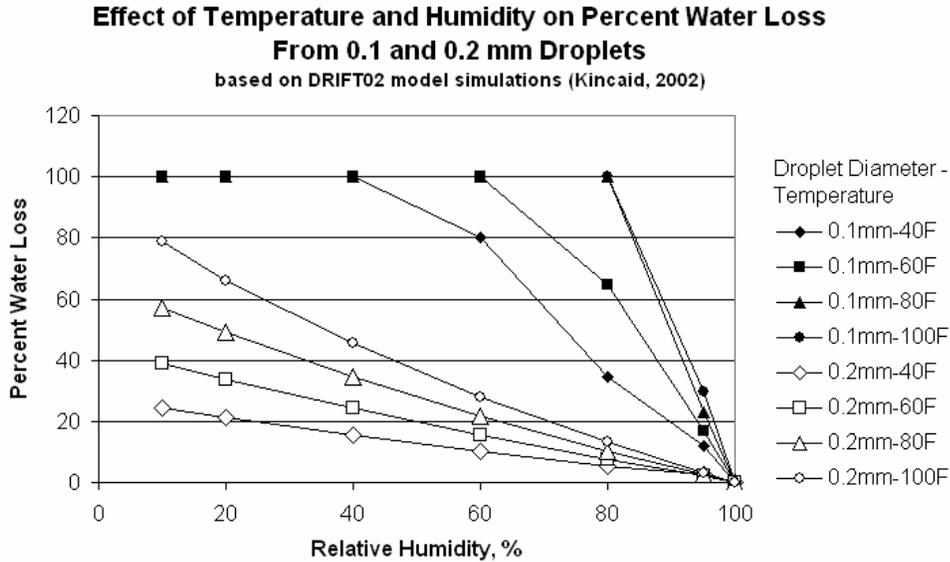


Figure 15. Effect of temperature and humidity on percent water loss from 0.1 and 0.2 mm droplets.

6.2.3 Mist and Aerosolization Fractional Efficiencies Table

The DRIFT02 analysis of the aerosolization process, described above suggests that 0.1 mm droplets are usually aerosolized during daytime conditions but may not be aerosolized when the humidity rises above 80 or 90% and the temperature decreases.

Under such conditions, the 0.1 mm fraction in Kincaid’s DRIFT02 database provides a reasonable estimate of aerosolization efficiency. In addition, transport of fine mist droplets 0.2 mm or larger, if any, occurs with very little change in size and never results in aerosolization under most northwestern growing season conditions.

This analysis suggests that the fractions of total sprayed flow that reside in the 0.1 mm and 0.2 mm size categories in Kincaid’s test database are useful for representing aerosolization and fine mist potential of various sprinkler configurations in the fate/transport model. The 0.1 mm and 0.2 mm categories

from Kincaid's database are provided in Table 5. The sprinkler designations, from Kincaid, typically represent the manufacturer (e.g. NS for Nelson, SEN for Senniger, RB for Rainbird), followed by the nozzle type and the orifice dimension in fractional inches. The relative dispersion and deposition behavior of droplets in these two size categories and larger sizes are evaluated in the dispersion model described in the next section.

Table 5. Aerosolization and fine droplet fractional efficiencies.

Fraction of Wastewater in Aerosolizable (0.1mm) and Mist (0.2mm) Droplet Sizes for Various Sprinkler Configurations									
Kincaid Test No. (a)	Sprinkler Nozzle Identification	Pressure (psig)	Fraction of Wastewater in Droplets with Initial Diameters equal to:		Kincaid Test No. (a)	Sprinkler Nozzle Identification	Pressure (psig)	Fraction of Wastewater in Droplets with Initial Diameters equal to:	
			0.1 mm (b)	0.2 mm (c)				0.1 mm (b)	0.2 mm (c)
81	'8070 11 psi Pam	11	0.0003	0.0010	17	'NSF85-sb 5/8	60	0.0017	0.0050
80	'8070 11psi water	6	0.0003	0.0010	71	'NS-SPFLAT 1/4	10	0.0017	0.0050
14	'NSF85-sb 1/2	30	0.0003	0.0010	76	'NS-SPFLAT 1/4	20	0.0017	0.0050
11	'NSF85-sb 3/8	30	0.0003	0.0010	69	'NS-SPFLAT 3/16	10	0.0017	0.0050
16	'NSF85-sb 5/8	30	0.0003	0.0010	26	'RB30-CD 5/32	40	0.0017	0.0050
54	'NS-SPCCMG 1/8	10	0.0003	0.0010	30	'SEN-WOBB-1/8	40	0.0017	0.0050
28	Sen Wobbler 1/8	10	0.0003	0.0010	13	'NSF85-sb 3/8	90	0.0020	0.0060
37	'SEN-WOBBLA-1/4	10	0.0003	0.0010	47	'NS-ROT D6 1/4	15	0.0020	0.0060
34	'SEN-WOBBLA-1/8	10	0.0003	0.0010	73	'NS-SPFLAT 1/4	30	0.0020	0.0060
35	'SEN-WOBBLA-1/8	20	0.0003	0.0010	78	'NS-SPFLAT 1/8	20	0.0020	0.0060
15	'NSF85-sb 1/2	90	0.0007	0.0020	75	'NS-SPFLAT 3/8	30	0.0020	0.0060
42	'NS-ROT D4 1/4	15	0.0007	0.0020	6	'RB30-sb 5/32	44	0.0020	0.0060
40	'NS-ROT D4 3/16	15	0.0007	0.0020	63	'SEN-LDN2P 1/4	30	0.0020	0.0060
56	'NS-SPCCMG 1/4	10	0.0007	0.0020	19	'NSF32-FCN 4GPM	60	0.0023	0.0070
58	'NS-SPCCMG 21/64	10	0.0007	0.0020	49	'NS-ROT D6 3/8	30	0.0023	0.0070
74	'NS-SPFLAT 3/8	10	0.0007	0.0020	53	'NS-ROTD6SPIN1/4	30	0.0023	0.0070
1	'RB30-sb 1/8	20	0.0007	0.0020	9	'RB30-sb 7/32	60	0.0023	0.0070
61	'SEN-LDN1P 1/8	30	0.0007	0.0020	64	'SEN-LDN3P 5/16	10	0.0023	0.0070
29	'SEN-WOBB-1/8	20	0.0007	0.0020	43	'NS-ROT D4 1/4	30	0.0027	0.0080
18	'NSF32-FCN 4GPM	30	0.0010	0.0030	48	'NS-ROT D6 1/4	30	0.0027	0.0080
20	'NSF32-FCN 5GPM	30	0.0010	0.0030	46	'NS-ROT D6 3/16	30	0.0027	0.0080
45	'NS-ROT D6 3/16	15	0.0010	0.0030	68	'NS-SPFLAT 1/8	30	0.0027	0.0080
52	'NS-ROTD6SPIN1/4	15	0.0010	0.0030	77	'NS-SPFLAT 3/16	20	0.0027	0.0080
59	'NS-SPCCMG 21/64	30	0.0010	0.0030	4	'RB30-sb 9/64	58	0.0027	0.0080
72	'NS-SPFLAT 1/4	15	0.0010	0.0030	22	'NSF32-FCN 5GPM	60	0.0030	0.0090
66	'NS-SPFLAT 1/8	10	0.0010	0.0030	55	'NS-SPCCMG 1/8	30	0.0030	0.0090
5	'RB30-sb 5/32	30	0.0010	0.0030	3	'RB30-sb 1/8	58	0.0030	0.0090
31	'SEN-WOBB-1/4	10	0.0010	0.0030	7	'RB30-sb 5/32	58	0.0030	0.0090
32	'SEN-WOBB-1/4	20	0.0010	0.0030	36	'SEN-WOBBLA-1/8	40	0.0030	0.0090
38	'SEN-WOBBLA-1/4	20	0.0010	0.0030	44	'NS-ROT D4 1/4	45	0.0033	0.0100
21	'NSF32-FCN 5GPM	40	0.0013	0.0040	51	'NS-ROTD6SPIN3/1	30	0.0033	0.0100
41	'NS-ROT D4 3/16	30	0.0013	0.0040	70	'NS-SPFLAT 3/16	30	0.0033	0.0100
57	'NS-SPCCMG 1/4	30	0.0013	0.0040	10	'RB30-sb 7/32	80	0.0037	0.0109
67	'NS-SPFLAT 1/8	15	0.0013	0.0040	8	'RB30-sb 5/32	72	0.0040	0.0120
25	'RB30-CD 5/32	30	0.0013	0.0040	23	'RB30-CD 9/64	30	0.0050	0.0150
27	'RB30-CD 11/64	30	0.0013	0.0040	65	'SEN-LDN3P 5/16	30	0.0050	0.0149
2	'RB30-sb 1/8	40	0.0013	0.0040	60	'SEN-LDN1P 1/8	10	0.0060	0.0179
33	'SEN-WOBB-1/4	40	0.0013	0.0040	62	'SEN-LDN2P 1/4	10	0.0063	0.0189
39	'SEN-WOBBLA-1/4	40	0.0013	0.0040	79	'4? field 11/64	20	0.0067	0.0199
12	Nelson 85 3/8	60	0.0017	0.0050	50	'NS-ROTD6SPIN3/1	15	0.0072	0.0218
					24	Rainbird 30 5/32	20	0.0138	0.0413

Notes: (a) From Kincaid DRIFT02 model files, personal communication, 2002. (b) Equivalent to Aerosolization Efficiency or Fraction Aerosolized, defined as undergoing 100% water evaporation under most conditions (i.e. temperatures above 60 deg F and humidities below 60% RH. (c) Fraction Mist in a size, 0.2 mm diameter, that is too large to aerosolize but may evaporate partially and drift in high wind conditions

6.3 Dispersion and Deposition Modeling

The atmospheric dispersion equations are defined in Section 3.4, and a range of simple and refined dispersion models suitable for microbial risk modeling are discussed in Section 3.4.3, page 26. An atmospheric modeling practitioner may use any of these models to estimate pathogen doses downwind of an irrigation system.

This section describes DEQ's development of refined, but practical modeling procedures, including the preliminary modeling studies conducted to determine how to best simplify the spray irrigation problem, avoiding droplet and aerosolized pathogen sizes that are not important to the dispersion and deposition outcome.

6.3.1 Dispersion and Deposition Behavior vs Aerosol/droplet Size

To simplify the dispersion work so that it is amenable to a spreadsheet modeling format, an analysis was conducted to determine how to treat each potential droplet or aerosol size category. The specific question to be addressed is, "What size droplet/particle contributes to significant concentration and deposition levels at or beyond a typical buffer zone distance of 300 ft (or 91m)?" Droplets in a fine spray may be up to 0.8 mm, and, after evaporation from a 0.1 mm drop, aerosolized particles may be as small as 10-50 μm in diameter, depending on the total solids content of the wastewater.

The ISC3 dispersion model was used to simulate both concentration and deposition from a hypothetical circular area source, such as an end gun spray pattern. The wind speed was 10 m/s and stability was neutral. The predicted concentration results for a unit emission source are presented graphically in Figure 16.

**Air Concentration vs Droplet Size at 10 mps Wind Speed
Unit emission for all sizes (1 cfu/s)**

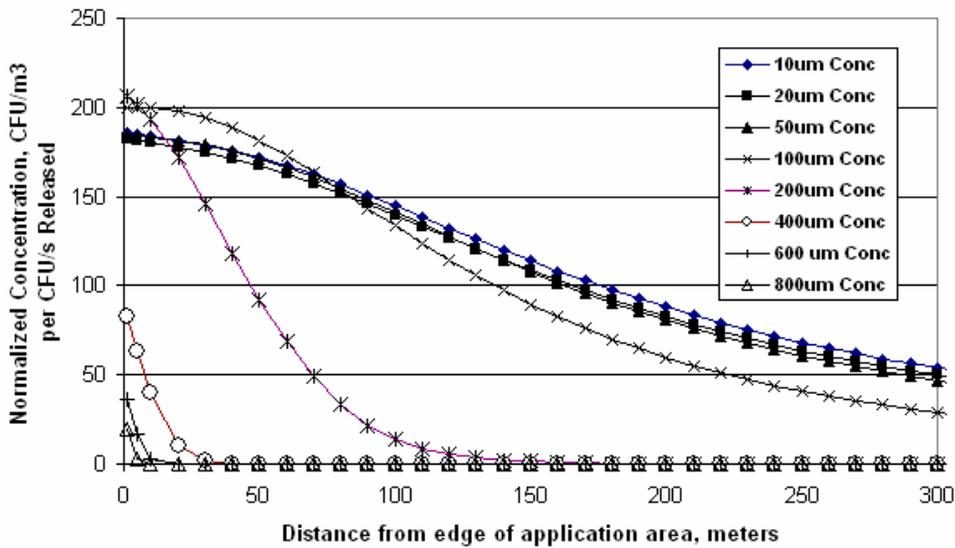


Figure 16. Air concentration versus droplet size with high wind speed

As indicated, all droplets larger than 400 μm drop out rapidly, with no influence, even under high winds, beyond about 30 meters. The 200 μm droplets have some minor influence out to a typical buffer zone distance of 300 ft (91 m). The 100 μm droplets travel well beyond the 300 ft distance.

The 10, 20 and 50 μm particles, which would be aerosolized particles whose size is dependent on the solids content of the wastewater, all behave similarly and can be assumed to travel the farthest, with relatively little settling.

In view of this analysis, all aerosolized particles will be given 20 μm size in future modeling, and they can be assumed to transport with only minimal settling out on surfaces.

6.3.1.1 High-wind droplet drift analysis.

An additional analysis was conducted, using only the droplet sizes available in the Kincaid database, to determine which droplets can be neglected because they do not travel a significant distance from the spray field, even under high wind conditions. This modeling analysis was conducted at a high wind speed (10 mps). The relative contributions to the total airborne microbial concentrations are shown in Figure 17. The relative contributions to surface deposition are shown in Figure 18 from the same model runs at 10 mps.

Contribution to Air Concentration vs Droplet Size
D Stability, 10 m/s Wind Speed

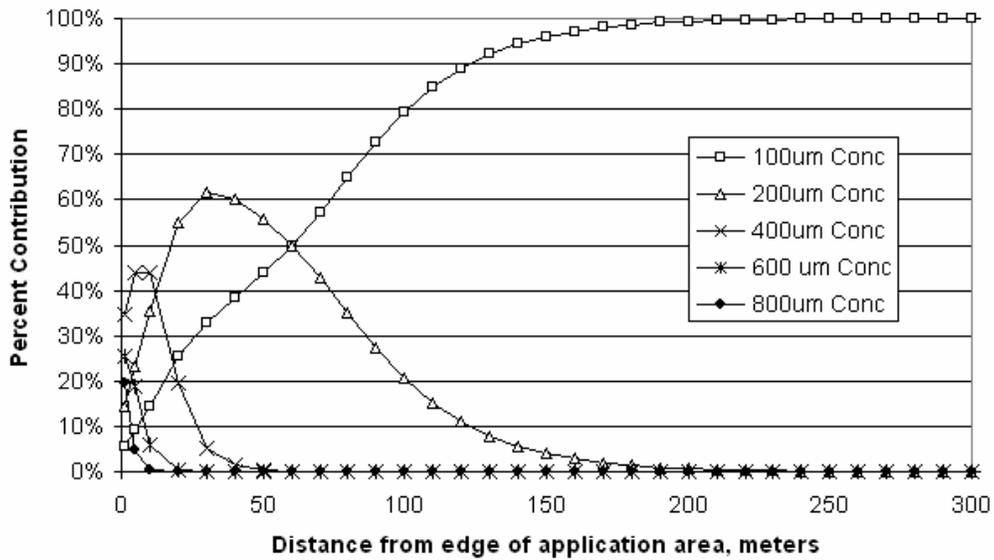


Figure 17. Contribution to air concentration versus droplet size.

Percent Contribution to Surface Deposition vs Droplet Size
D Stability, 10 m/s Wind Speed

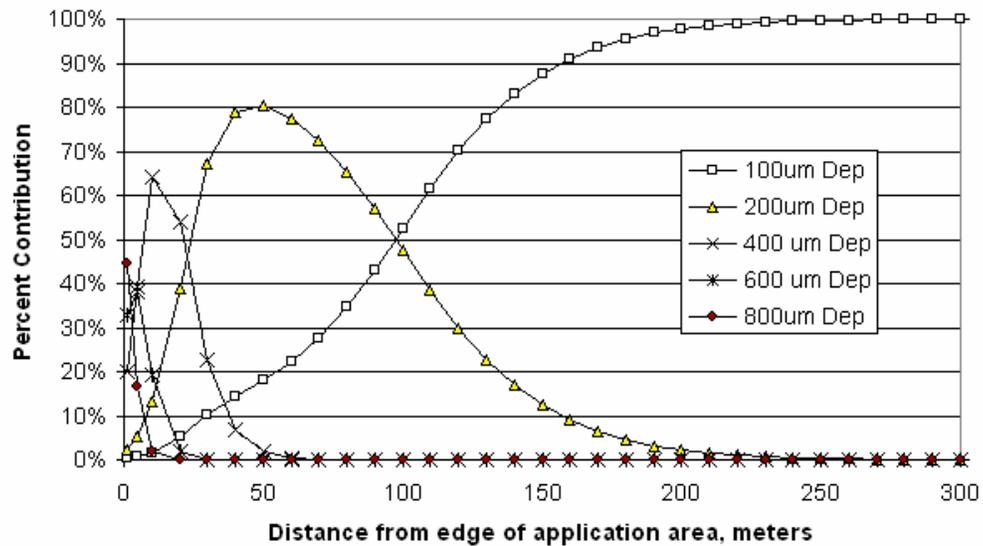


Figure 18. Percent contribution to surface deposition versus droplet size.

As seen in Figure 18, the 0.2 mm droplets (200 μm) may play a relatively small but still significant role in both concentration estimates and deposition estimates beyond 100 m under high wind conditions. The 0.1 mm droplets are very significant contributors to both concentration and deposition at all

distances, while 0.2 mm droplets are significant at this high wind speed only out to about 150 m.

Following repeated use of this model on actual applications, it may be concluded that the 200 μm droplets play a minor role in microbial risk estimates, and their use in the model may be discontinued. However, at this stage of development, wind-driven mist droplets should be evaluated in addition to the aerosolized components until a better understanding of their risk contribution is available.

Based on this preliminary analysis, the microbial fate/transport model will be configured to address both 0.1 mm and 0.2 mm droplet dispersion and deposition, as well as aerosolized particles with a size of 20 μm in diameter. This is an approximate size of a dried residual aerosol particle resulting from a 100 μm droplet containing wastewater with a total solids content of about 6,000 mg/l.

In summary, the modeled concentration and deposition estimates generated for the DEQ MIRA spreadsheet model involve only the 20 μm , 100 μm and 200 μm size categories. Since the 20 μm aerosol particles represent desiccated 100 μm droplets, and since the desiccation power of the atmosphere changes rapidly on an hour to hour basis, either one may be appropriate, but not both at the same time. It was therefore decided to conduct dispersion estimates for both 20 μm aerosols and 100 μm droplets and to utilize the concentration and deposition estimates for whichever of these two size categories result in the largest concentration and deposition at each downwind distance.

6.3.2 Incorporating Refined Dispersion Modeling by Component

The ISC3 model, described in Section 3.4.3.3, page 28, is one dispersion model alternative that is most appropriate for refined modeling purposes. It may be used to simulate a single sprinkler or a simple configuration of irrigation system components by adding dispersion results from separately modeled components—the endgun and rotators from a pivot irrigation system, for example. For this configuration, a single end gun and a line of rotators are modeled separately to provide worst case concentrations for each component. Dispersion modeling results are generated in a normalized fashion using unit emission rates (e.g. the dispersion factors D_d have the units CFS per cubic meter per CFS/s emission rate). The dispersion and deposition factors are imported into the MIRA spreadsheet for both component (e.g. endgun and rotator) simulations. Results at receptor distances located every 10 meters

downwind from the edge of the spray area (= 0 m) out to 1,000 m downwind are used. The calculations of concentration and deposition amounts, die-off and risk are then conducted at each distance so that microbial risk may be estimated every 10 m out to 1 km.

6.3.3 Worst Case Dispersion Outputs, With and Without Decay

The dispersion factors generated by the refined modeling for an end gun are shown in Figure 19. These curves are independent of the source term, but rather, reflect the product of the dispersion factors, D_d times the decay factor, M_d (in this case for fecal coliform).

The upper curves represent the maximum predicted dispersion factor for day and nighttime conditions assuming no biological decay.

The lower two curves represent the maximum dispersion conditions for day and night with the decay factor, M_d factored in to represent the combined $D_d \times M_d$ product. This product, multiplied by the source term (Q_s) represents the predicted increase in pathogen concentration. See Equation 1, page 18.

Thus, the values here could be used to make worst case estimates for an endgun configuration if the source term quantities are known. These curves would over-predict any larger application area, such as a pivot or rectangular field, since an end gun is virtually a point source. It should be noted that most daytime conditions will result in curves that lie below the daytime-with-decay (bottom curve). It is also clear that nighttime dispersion results in significantly higher concentrations, as indicated by the dotted-line curves in comparison to the solid black, daytime curves.

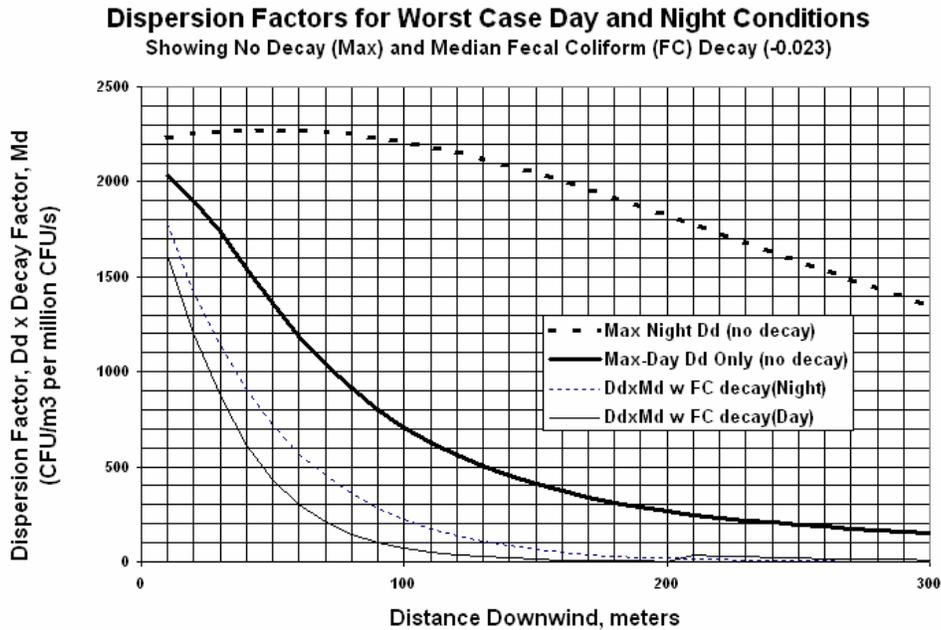


Figure 19. Dispersion factors for worst case day and night conditions.

6.4 Risk Estimates

The microbial risk procedures outlined in Section 5 of this report have been incorporated into a parallel portion of the MIRA spreadsheet to provide internal preliminary estimates of the risk of infection at various receptor distances downwind. The inhalation pathways and the deposition-on-produce pathway are both estimated and summed to provide a total dose. As described in Section 5, inhalation doses are summed as ingestion doses because the droplets and aerosol particles are both large enough to be removed before reaching the lungs, cleared by the mucociliary processes and swallowed.

6.5 Model Calibration and Other Needs

Field studies can be performed to verify and/or provide additional impact information to evaluate impacts. It is important to monitor conditions that affect atmospheric concentrations of microorganisms, including the following:

- Meteorological data for each time step, including wind speed, wind direction, temperature, cloud cover, and relative humidity.
- Application rate of material land-applied.
- Representative microorganism content of material land-applied.

- Sampling method used, including growth media used and types of microorganisms monitored.

Field calibration and validation of microbial drift/deposition modeling is important when comparing computer modeled microbial deposition / aerosolized microbial density results with field measurements. For budgetary reasons, this activity should be limited to characterizing deposition onto deposition plates – an inexpensive analysis – to characterize both deposition and microbial densities indirectly. Depending upon the correlations obtained in field studies, modeling parameters can be modified and the model calibrated to actual Idaho field conditions.

It would be an important extension of the modeling effort to consider the development of stand-alone pre-run model output tables. Wastewater source term microbial characterization could be utilized in drift / deposition modeling. Modeling would be done on representative wastewaters utilizing typical irrigation systems and varying meteorological conditions. Microbial density estimates for site-specific scenarios would be pre-modeled and provided to the regulated community. This would include graphs and nomographs for facilities to consult to determine the appropriate irrigation system design, given site specific meteorological conditions and wastewater characteristics, to design and operate wastewater land treatment systems without the need for time consuming and expensive site-specific modeling.

6.6 Comparison of Modeled vs. Field Dispersion Studies

Evaluation of the dispersion/biological decay model requires an extensive field program. Field studies published in the literature can be useful for comparison purposes, but the level of input data to accurately simulate a literature field study is difficult to find. One or more source data such as total flow rate, aerosolization efficiency, and wastewater pathogen loading data are typically missing. Nevertheless, by normalizing the modeled dispersion/decay curves against the maximum measured microbe concentration in a field study, the data can be generally compared to the shape and decay rate of the modeled curves. In this manner, comparison can only be made of the product of the dispersion factor, D_d times the biological decay factor, M_d . However, this partial comparison is a major component of the model and is worthy of evaluation, even if it is not a rigorous one.

Two studies were described in the literature review in Section 2 involving wastewater spray application. The data from those two studies have been

plotted in comparison to the normalized dispersion/decay curves existing in the DEQ model application described for a single endgun.

6.6.1 Parker et al (1977) Measurements

Parker, et al. (1977) studied bioaerosols from potato processing wastewater spray fields. Coliform bioaerosols from the spray field were collected 392 m from the source. Data from Parker’s eight field trials are plotted against the nighttime and daytime maximum concentration curves (no biological decay) and the maximum nighttime and daytime curves with median fecal coliform decay for comparison (Figure 20). The measured data seem to behave very similarly to the day and nighttime decay curves with median fecal coliform decay. The curves with no decay appear to severely over-predict the measured values.

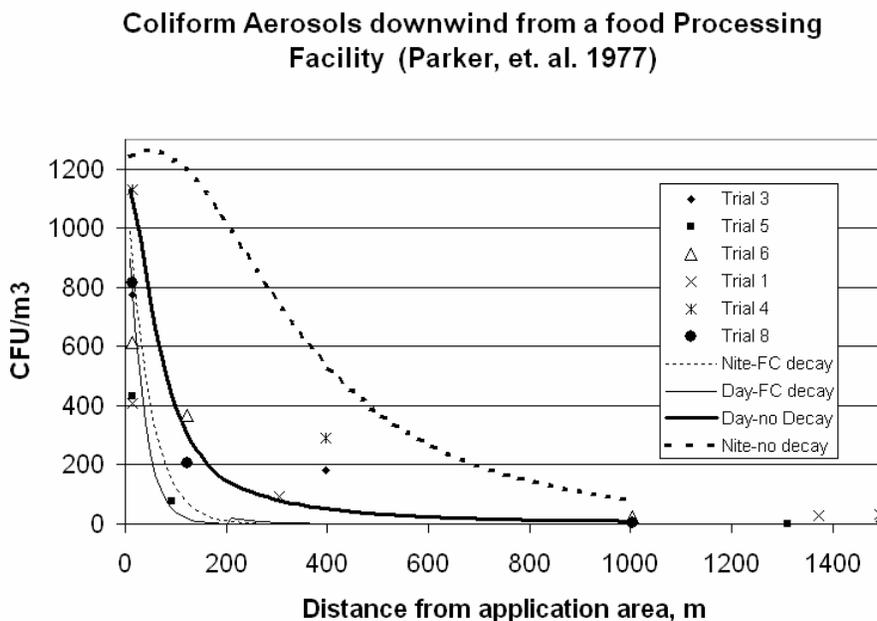


Figure 20. Coliform aerosols downwind from a food processing facility.

6.6.2 Katzenelson and Teltsch (1976) Measurements

Aerosolized enteric bacteria (coliform and salmonella) from a spray irrigated field near Kibbutz Tsorah, Israel were studied by Katzenelson and Teltsch (1976). Aerosolized coliform and salmonella bacteria were found 350 m and 60 m downwind respectively. Data from Katzenelson and Teltsch’s nineteen field tests are plotted against the nighttime and day-time maximum concentration curves (no biological decay) and the maximum nighttime and day-time curves with median fecal coliform decay for comparison (Figure 21). Again, the measured data seem to behave very similarly to the day and

nighttime decay curves with median fecal coliform decay. The day and night curves with no decay appear to over-predict and severely over-predict the measured values respectively.

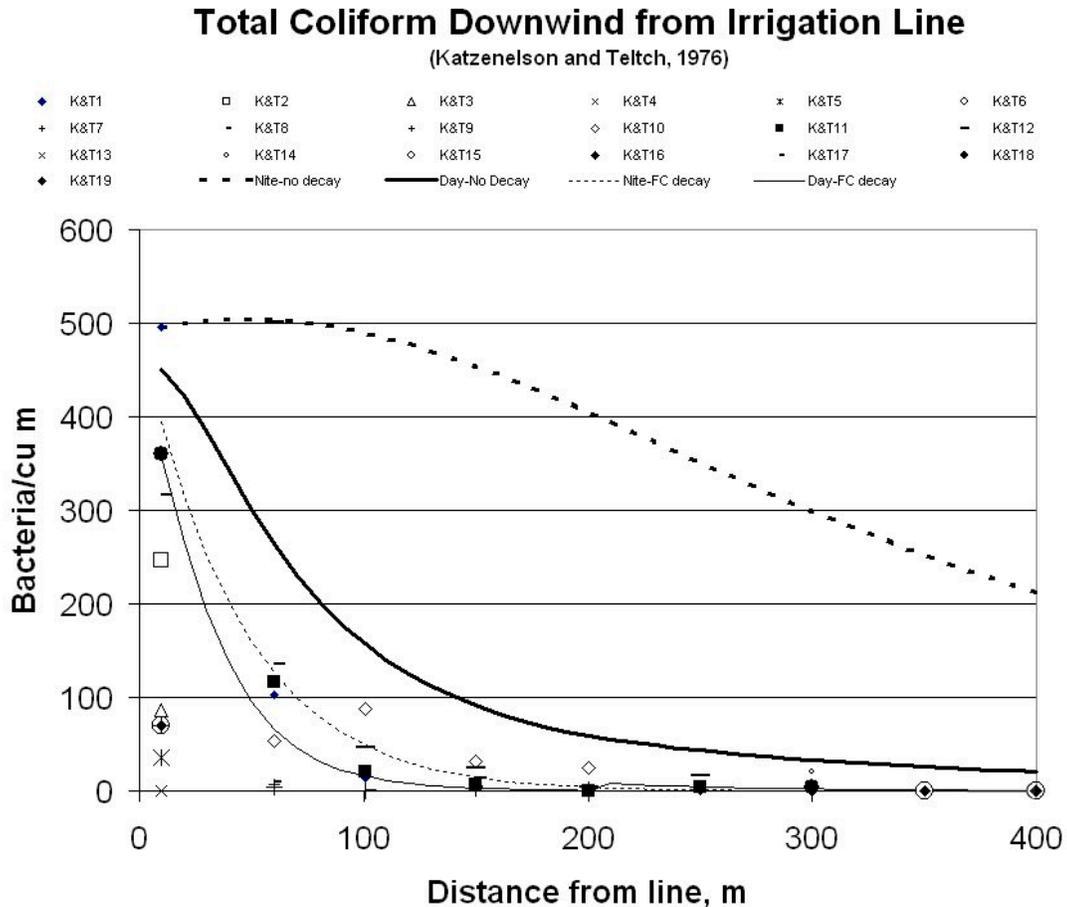


Figure 21. Total coliform downwind from irrigation line.

6.6.3 Conclusions from Limited Field Comparisons

The comparisons of modeled dispersion-decay curves to the data in these studies indicates that the data are generally bounded by the maximum dispersion curves when fecal coliform median biological decay term is included. Data that fall well below the “max day” and “max night curves were likely conducted during more typical (better) dispersion conditions. The maximum day and night dispersion curves without a decay term are generally much higher and do not reflect the bulk of the measurements. They should be considered extreme worst case dispersion results that are not likely to occur unless significant nighttime or rainy-daytime irrigation is occurring.

7. Example Land Application Analysis

An example Microbial Risk Analysis (MIRA) is presented in this section to provide some indication of typical results for a hypothetical land application operation.

DEQ has implemented the fate/transport/risk components, described in Sections 3 through 6 of this report, in a spreadsheet format that allows an expert user to conduct repeatable analyses in a reasonable time frame. However, there is need to develop both a more user-friendly version of this spreadsheet as well as a user manual.

The analysis described below should not be considered a unique computation. The same analysis could be conducted in a less refined or more refined manner. The analysis is moderately refined, but it is not conducted for a specific regulatory purpose, so some of the parameters with high amounts of uncertainty may be selected differently when conducting this analysis for a regulatory purpose (i.e., protection of public health).

7.0 Hypothetical Pivot Operation

A typical pivot operation is hypothesized, consisting of a 1,300 ft pivot arm bearing about 93 rotator type nozzles (D6 3/16) operated at 30 pounds per square inch (psi) pressure and assumed to provide uniform coverage along its entire length. The pivot arm terminates at an end gun (NSF 85) with a 5/8 inch orifice, operated at 60 psi.

The wastewater, containing one million Colony Forming Units (CFU) per 100 milliliters (1×10^6 CFU/100 ml) of *E. coli* is pumped to the system with 700 gallons per minute (gpm) distributed to the pivot rotator nozzles and 87 gpm distributed to the end gun. (Note: The infectivity model parameters used for *E. coli* in this example are based on best-fit dose response parameters from an enteric pathogen feeding study and may not be representative of the distribution of *E. coli* subtypes in different wastewater streams.)

This wastewater microbial loading is much higher than most systems, however it was selected to better illustrate the model results. The wastewater has a total solids content of 6,000 mg/l and solids have a density of 1 gm/cm³.

7.1 Aerosolization Efficiencies

Aerosolization efficiencies (fraction less than 0.1 mm) and the fine mist fraction (< 0.2 mm) are found in Table 5, page 65. The rotators best correspond to Kincaid test number 46 and the end gun best corresponds to Kincaid test number 17. The 0.1 mm and 0.2 mm fractions are .0027 and 0.008, respectively, for the rotators, and 0.0017 and 0.0050 for the end gun.

7.2 ISC3 Dispersion Modeling for Pivot Components

Dispersion modeling of the pivot arm rotators and the end gun was conducted separately using the ISC3 model. The pivot arm was modeled as a rectangular area, 1,300 feet long and 42 feet wide, with the wind aligned along the direction of the arm to be conservative. The nozzles are 4 feet above the ground. The end gun was modeled as a circular area, 164 feet in diameter with an 18 foot effective release height to account for a 12 foot sprinkler height and an upward droplet trajectory. The two separately modeled source areas are conceptually aligned with the end gun at the end of the pivot arm and downwind concentrations for each (every 10 meters) were input into the MIRA spreadsheet, where concentrations are mathematically superimposed.

Both concentration and deposition outputs were generated by the ISC3 model for 200 μm , 100 μm and 20 μm “particles.” (Note: 0.2 mm = 200 μm , and 0.1 mm = 100 μm .) The 20 μm size is modeled for the desiccated (aerosolized) particle because the 100 μm droplets containing 6000 mg/l solids are estimated in the MIRA spreadsheet to have a diameter of 18 μm after the water has evaporated. This size also has implications in the risk analysis, because any particles this size will be removed in the nasopharyngeal region, cleared by the mucociliary escalator, and swallowed. This fact simplifies the risk analysis because now all doses, both from inhalation and from ingestion of produce, will result in an “ingestion” pathway. This is convenient since most of the dose-response data are for ingestion. The 100 μm and 20 μm contributions were not summed, but the maximum value of the pair was selected in the calculation because droplets may aerosolize or not aerosolize at any time, depending on humidity and temperature conditions.

Five sets of hourly meteorological conditions were modeled: *neutral*, or “D” Stability with 2.5, 5, and 10 meters per second wind speeds; Unstable daytime conditions, or “B” stability with 1 mps wind; and very stable nighttime conditions, or “F” stability with 1 mps winds. The neutral stabilities reflect typical daytime windy conditions, while unstable conditions occur only when

winds are light and the sun is shining. Very stable conditions represent worst case dispersion conditions. It should be noted that light wind/stable conditions allow the larger droplets to settle rapidly but cause the airborne concentrations to be the highest, while the high wind neutral conditions result in lower airborne concentrations, and maximum deposition at more distant receptors.

ISC3 model results are output as one-hour averages. For “event” or “daily” calculations, an 8-hour average is assumed, as this is the most the wind typically blows toward one receptor in a day. A 1-hour to 8-hour averaging period conversion factor of 0.7 is applied to “daily” or “event” based calculations as described in Section 3.4.4.6 of the report. For annual average doses, the 1-hour to annual conversion factor of 0.08 is applied.

7.3 Fate and Transport Parameters

Microbe impact and biological decay (viability) factors are obtained in Table 1 (page 22) and Table 2 (page 23) respectively, Section 3. Values for “Total Coliforms” are used to represent *E. coli*. For typical D stability conditions, the median values are used ($I = 0.34$, $\lambda = -0.032$). For daytime unstable conditions (B stability) the 40th percentile values are used to reflect high UV, low humidity effects ($I = 0.27$, $\lambda = -0.05$). For nighttime stable conditions, the 60th percentile values are used ($I = 0.48$, $\lambda = -0.020$) to reflect high humidity, low-UV, conditions amenable to less biological decay.

7.4 Risk Parameters

The Beta-Poisson dose-response model is used for *E. coli* with alpha and beta coefficients of 0.1705 and 1.61E+06 respectively from Table 4, page 40, Section 4.2. Risk-related parameters for the inhalation and ingestion-from-produce pathways are shown in Table 6.

Table 6. Risk analysis parameters used in hypothetical example.

Parameter	Value	Units
Inhalation Parameters		
Breathing rate	20	m ³ /day
Event duration	8	hrs
Total Volume inhaled	6.67	m ³
Fraction of inhaled aerosol ingested	1	unitless
Aerosol diameter	18	micrometers
Ingestion on Produce Parameters		
Daily produce ingestion rate, IR _p	40	g/day (dry wt)
Fraction of produce from home, F _h	0.8	unitless
Number of produce meals per year, q	350	meals/yr
Interception fraction, R _p	.39	unitless
Plant surface loss coefficient, K _p	18	yr ⁻¹
Length of plant exposure per harvest, T _p	.164	yr
Land application event duration, ED _L	8.0	hr
Land application event frequency, EF	25	yr ⁻¹
Yield of edible standing crop biomass, Y _p	2.24	kg dry wt/m ²

7.5 Dispersion Modeling Results

Preliminary dispersion modeling results for 5 sets of meteorological conditions are shown in Figure 22, Figure 23, and Figure 24. These three graphs depict the airborne concentration, the surface deposition rate, and the total dose (which includes both the airborne inhalation pathway and the ingestion of produce.) Microorganism biological decay is included in these graphs along with dispersion. Each graph depicts the result for each of the five modeled meteorological conditions.

In Figure 22, depicting airborne concentrations, the nighttime F stability conditions result in the highest concentrations due to the low wind speed and minimal dispersion. High-wind conditions result in lower airborne concentrations because of the dilution effect of the higher winds. Low-wind daytime conditions (B stability, 1 mps) result in low concentrations because the dispersive effect of the atmosphere is the greatest.

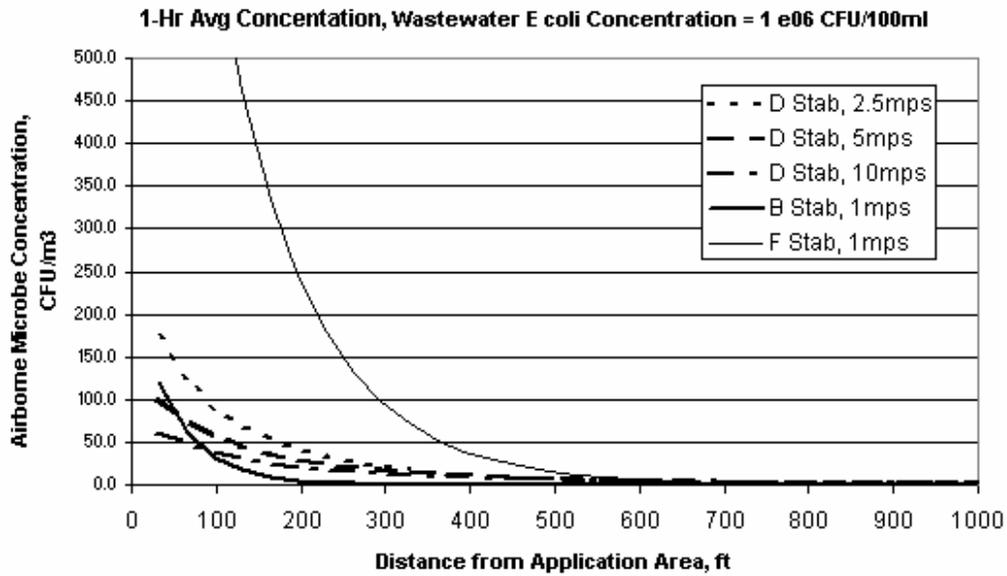


Figure 22. Airborne microbe concentration versus atmospheric conditions. Low wind nighttime (F stability) conditions are the worst case for the inhalation pathway.

In Figure 23, depicting the surface deposition rate, the high-wind conditions result in the greatest deposition rates because larger droplets (200 μm) are carried greater distances from the application area. As with the airborne concentration graph, neutral conditions, with low to moderate winds, result in intermediate deposition levels.

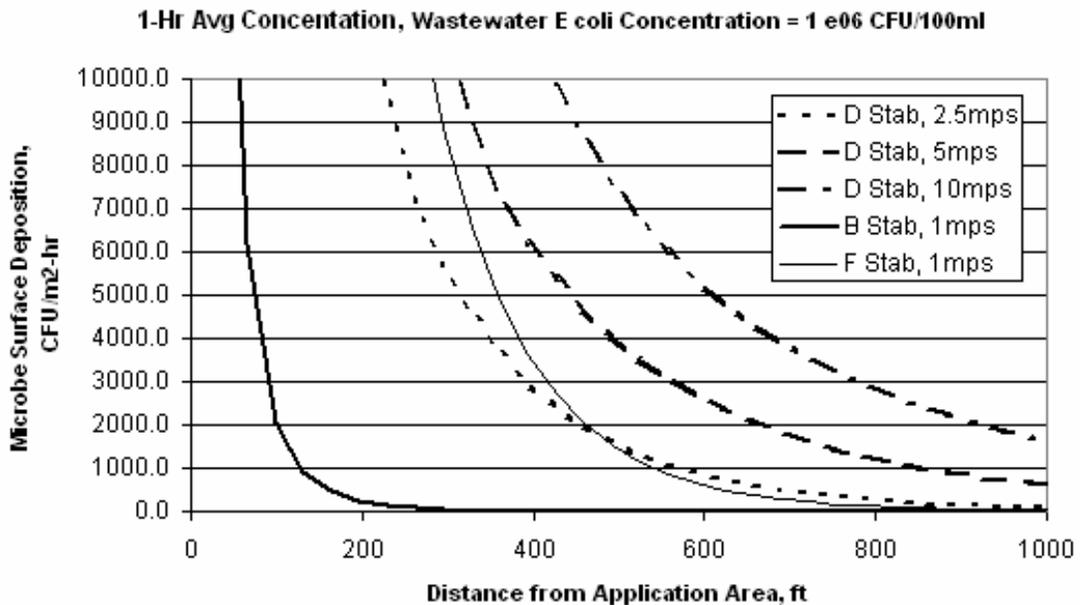


Figure 23. Microbe surface deposition versus atmospheric conditions. High wind conditions are the worst case for the deposition/produce consumption pathway. This shows the combined effect of airborne and deposited/ingested microbes on total dose.

In Figure 24, the total dose is depicted. Total dose represents the combined doses from inhalation/ingestion of the airborne microbes and from ingestion of microbes deposited on produce, which is then ingested. In this case, it can be seen that the stable nighttime conditions (F stability) predominate at distances closer than about 350 feet, while the high wind conditions (D stability, 10 mps) cause the highest dose beyond that point. Thus, inhalation/ingestion is important near the source while deposition/ingestion is more important further away. Typical daytime conditions (D stability, 2.5 mps) again result in intermediate doses.

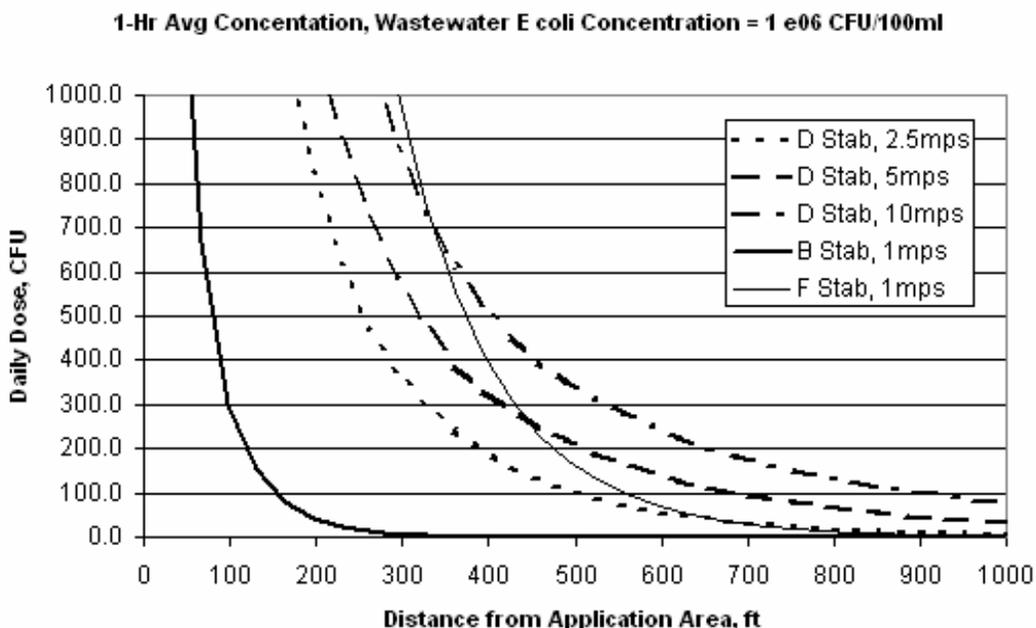


Figure 24. Total dose versus atmospheric conditions. This shows the combined effect of airborne and deposited/ingested microbes on total dose.

The relative contributions to total dose that results from the stable, (F) worst case conditions and high-wind worst case conditions can be seen in Figure 25 and Figure 26, respectively. Both graphs depict approximately the same total dose levels. However, for the stable conditions shown in Figure 24, inhalation/ingestion of the airborne (suspended) bioaerosol is the largest contributor to total dose, while, for the high-wind conditions shown in Figure 25, ingestion of produce with surface-deposited microbes represents the predominant pathway.

Pivot with 1 million CFU/100ml at 787 gpm

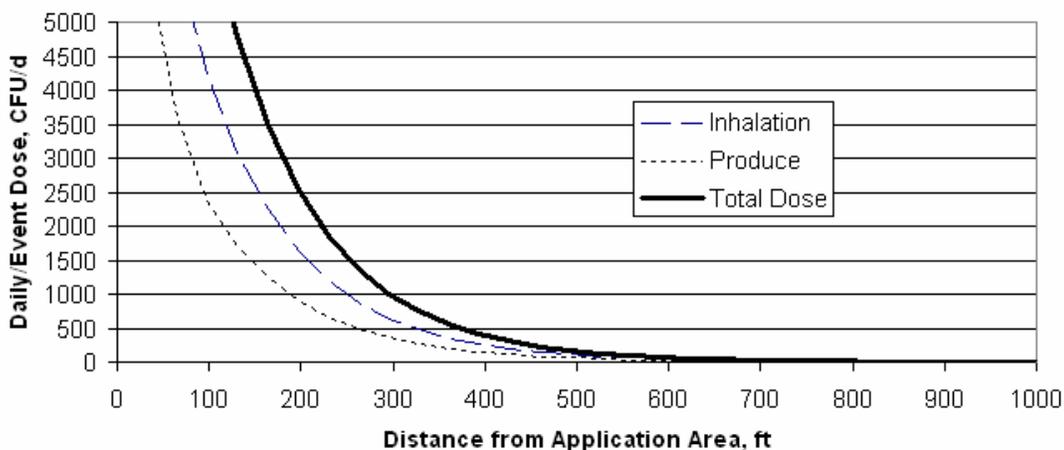


Figure 25. Dose contributions for E. coli: F Stability, 1.0 mps. Inhalation is the predominant pathway under nighttime, very stable, low wind conditions.

Pivot with 1 million CFU/100ml at 787 gpm

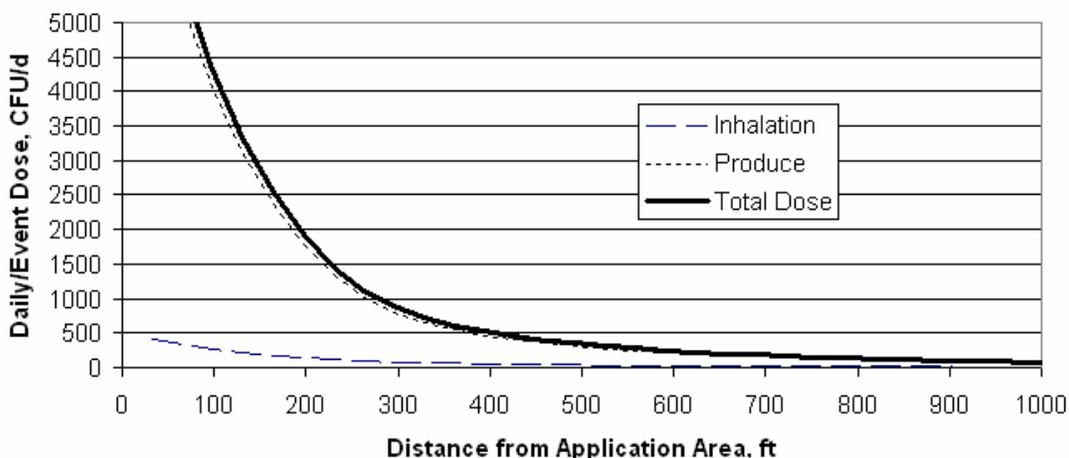


Figure 26. Dose contributions for E. coli: D Stability, 10.0 mps. At high wind speeds, deposition on produce, followed by ingestion, contributes the greatest dose.

The results described in the following section (Risk Analysis Results) represent the most typical atmospheric conditions (D stability, 2.5 mps), but not necessarily the most conservative. If nighttime irrigation is conducted, the nighttime (F) conditions should be used. If daytime-only operation is planned, the high wind conditions may be the most conservative. The averaging period conversion factors (Section 3.4.4.6) assume that the maximum one-hour results are used, and serve to obtain appropriate long-term averages. Thus, for a conservative analysis, the set of meteorological conditions causing the maximum total dose should be used.

7.6 Risk Analysis Results

Risk results from the MIRA spreadsheet are described below.

7.6.1 Results for Hypothetical configuration at 10^6 CFU/100ml

Microbial Risk Analysis (MIRA) results are presented Figure 27, Figure 28, and Figure 29, representing the total dose, the daily risk or probability of infection, and the annual risk, respectively. As seen in Figure 27, with one million CFU/100 ml, the total dose at the traditional 300 foot buffer zone distance is about 350 CFU per day. The produce pathway is the largest contributing roughly about 200 CFU while the inhalation pathway contributes roughly about 150 CFU under these typical daytime conditions.

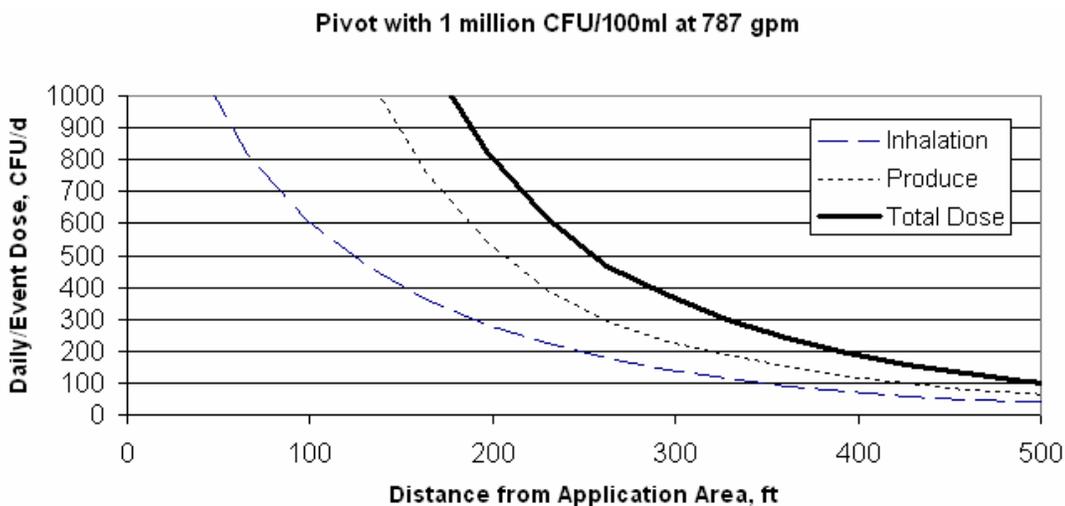


Figure 27. Dose contributions for E. coli: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions.

Figure 28 shows the daily risk of infection from inhalation and ingestion of produce. The total risk at 300 feet distance from the application area is approximately 4×10^{-5} , or one in 25,000. Produce contributes a slight majority of this risk as expected from the relative dose contributions discussed above.

FEBRUARY 2006
 Pivot with 1 million CFU/100ml at 787 gpm

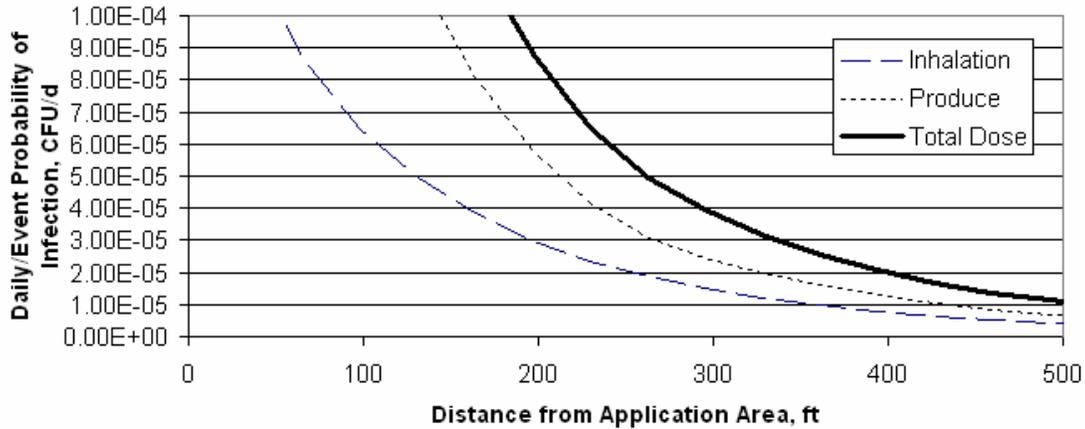


Figure 28. Daily risk contributions for E. coli: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions.

Figure 29 shows the annual risk of infection. The total annual risk is just over 1×10^{-3} , or one in 1,000 at 300 feet. This level of infection risk is elevated and some mitigation is probably warranted. The dose and risk levels over a wider range of distances can be seen in Figure 30, Figure 31, and Figure 32, which use a logarithmic scale. Daily total risks in the range 10^{-4} to 10^{-6} are found at 200 and 900 feet, respectively from the application area (Figure 31). Annual total risks equivalent to the 10^{-4} and 10^{-6} levels are found at 700 and 1,600 feet from the application area, respectively (Figure 32).

Pivot with 1 million CFU/100ml at 787 gpm

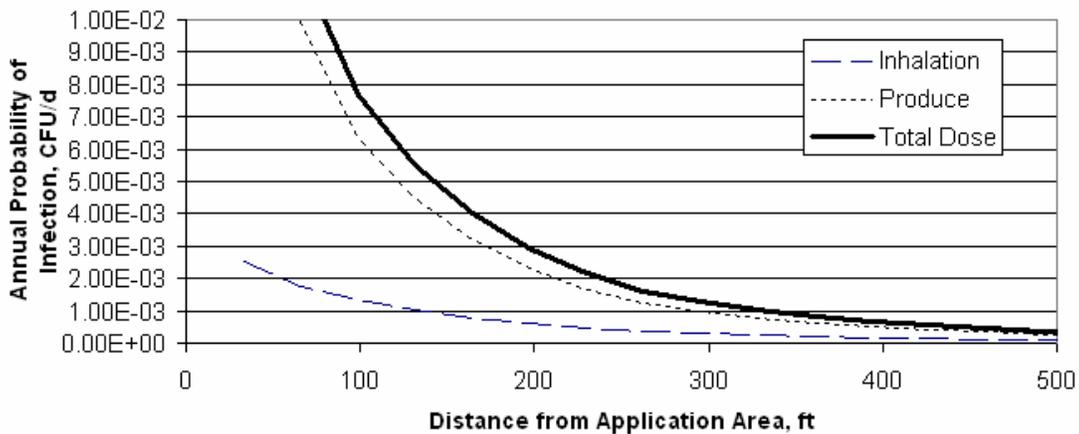


Figure 29. Annual risk contributions for E. coli: D stability, 2.5 mps. Standard model output for hypothetical system (10^6 CFU/100 ml) under typical daytime conditions.

Pivot with 1 million CFU/100ml in Wastewater at 787 gpm

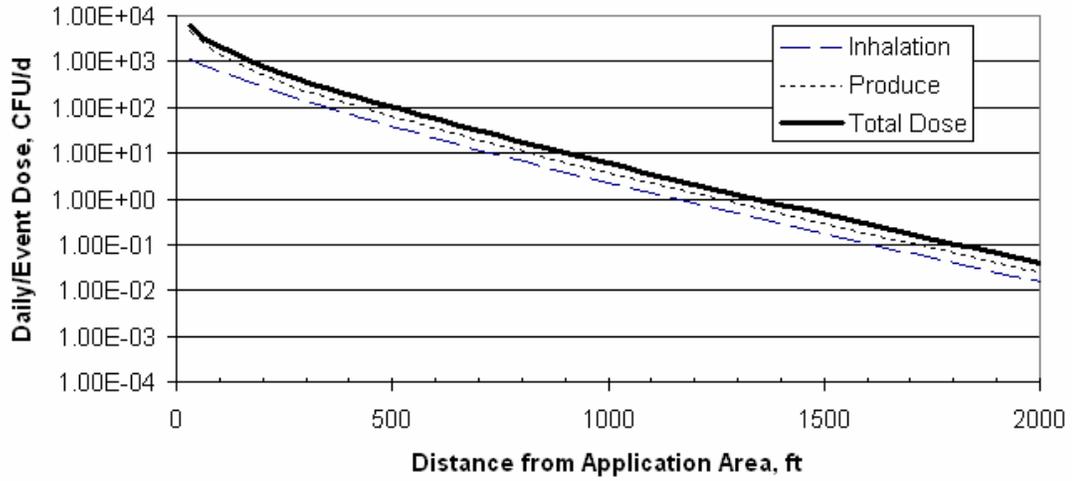


Figure 30. Dose contributions for E. coli: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and doses.

Pivot with 1 million CFU/100ml at 787 gpm

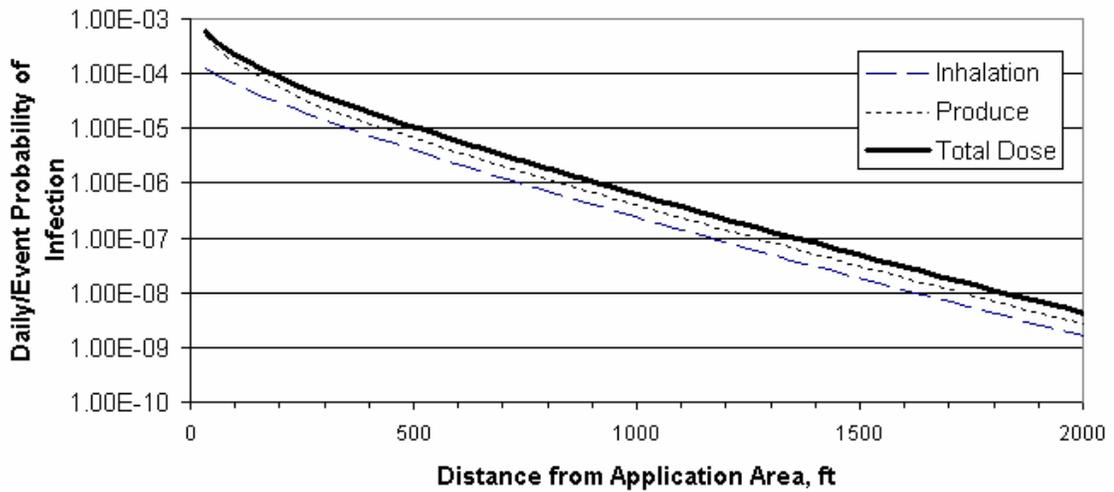


Figure 31. Daily risk contributions for E. coli: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and doses.

Pivot with 1 million CFU/100ml at 787 gpm

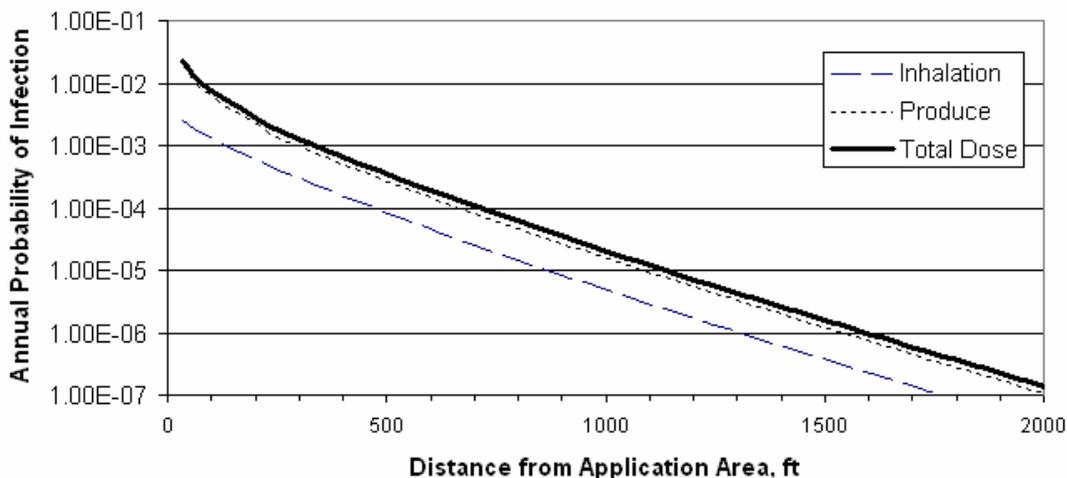


Figure 32. Annual risk contributions for E. coli: D stability, 2.5 mps. Logarithmic scale used to depict wider range of distances and annual doses.

This analysis shows that the buffer zone distances required to obtain nominally acceptable risk ranges are somewhat beyond the 300 foot standard buffer zone for this hypothetical scenario at 10^6 CFU/100 ml. This suggests that some reduction in microbial loadings or modification of the system configuration may be necessary to be acceptable in most locations.

The final model result output format allows exploration of alternative wastewater loadings. These graphs, presented in Figure 33-Figure 35, show decade reductions in wastewater loading and the relative daily dose, daily risk, and annual risk predictions. As seen in Figure 34, daily risks at 300 feet could be reduced to 10^{-6} (one-in-a-million) by lowering the wastewater loading to somewhere between 10,000 and 100,000 CFU/100 ml. Similarly, Figure 35 shows that an annual risk at 300 feet of 10^{-4} may be reached by simply reducing microbial loadings in the wastewater by one decade, to 100,000 CFU/100 ml. This level of risk is similar to levels considered by EPA for drinking water and may be acceptable by analogy here. However, if a jurisdiction wished to reduce annual risk of infection to 10^{-6} , the wastewater microbial loadings should be reduced to 1,000 CFU/100 ml or alternative system modifications may be appropriate.

FEBRUARY 2006

787 gpm Pivot

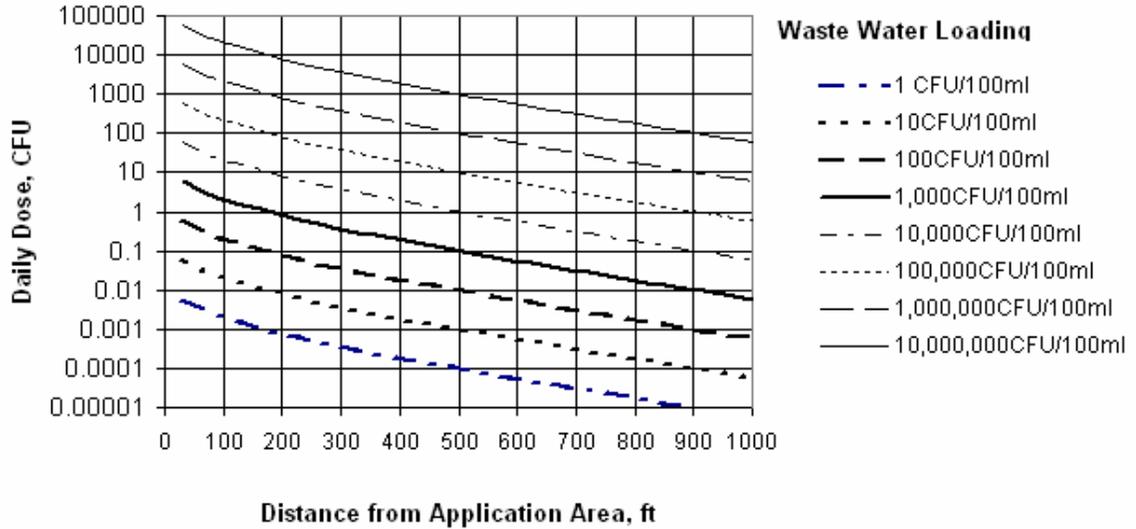


Figure 33. Variation in daily dose with distance, E. coli: D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings.

787 gpm Pivot

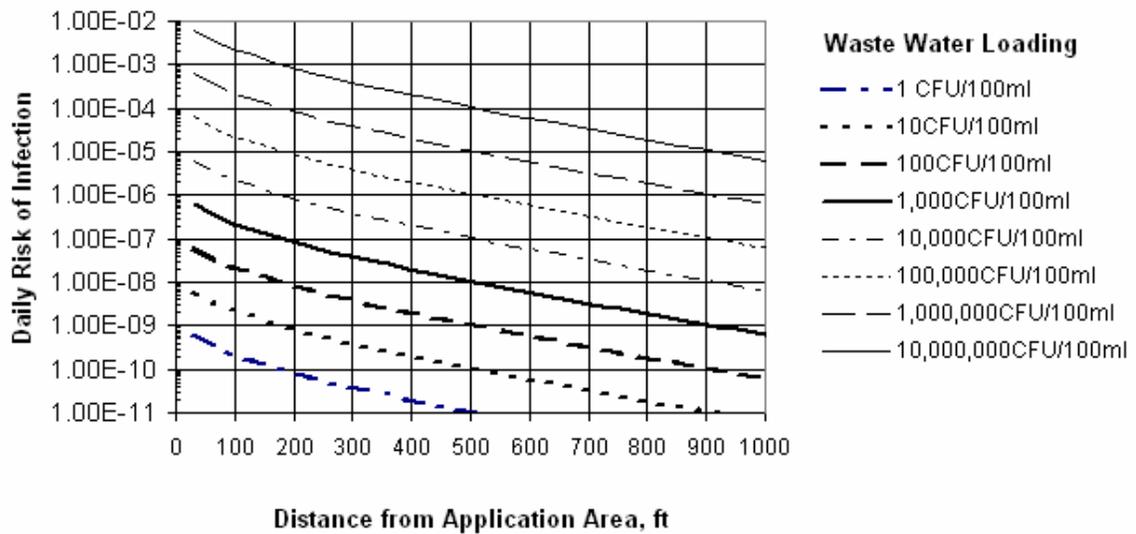


Figure 34. Variation in daily risk of infection, E. coli, D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings.

787 gpm Pivot

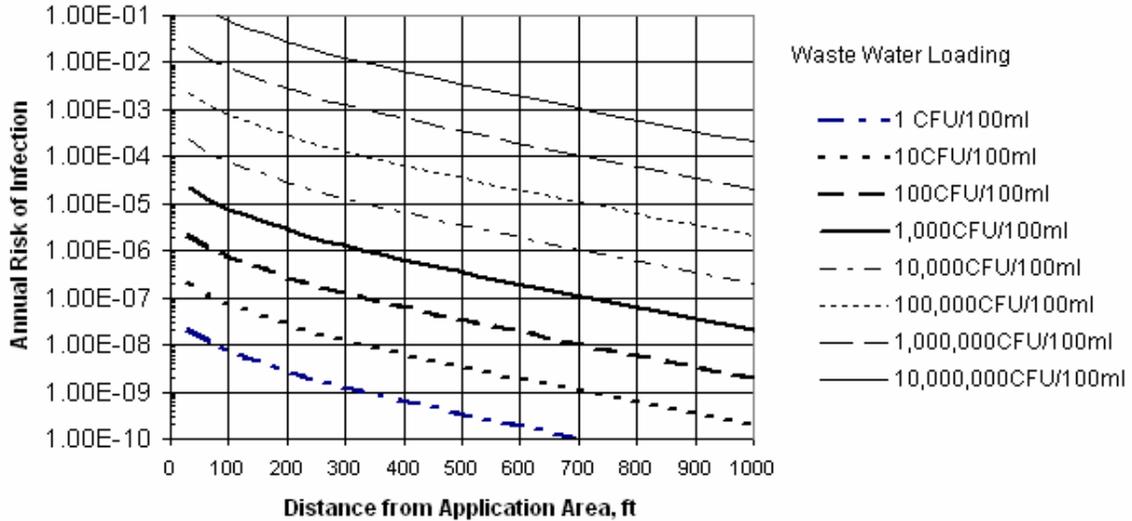


Figure 35. Variation in annual risk of infection, *E. coli*, D stability, 2.5 mps. Model results over a wide range of wastewater microbial loadings.

7.6.2 Results for Reduced Wastewater Loading, 1,000 CFU/100 ml

The hypothetical configuration discussed in the preceding was revised to reflect a three-decade reduction of wastewater *E. coli* concentrations from 10^6 to 10^3 (or 1,000 CFU/100 ml). Dose, daily risk and annual risk of infection graphs are shown in Figure 36, Figure 37, and Figure 38, respectively, for the MIRA analysis utilizing 1,000 CFU/100 ml. Figure 37 indicates that daily risk will be 4×10^{-8} , well below any level of concern, at the 300 foot buffer zone distance. The annual risk, depicted Figure 38 indicates that a risk of infection from *E. coli* is close to 10^{-6} , or one-in-a-million, at the 300 foot distance.

This level of risk would probably be acceptably conservative in most design scenarios. However, a three-decade reduction in wastewater loading may not be economically feasible. In this case, other design features in the hypothetical land application design should be reviewed, such as sprinkler selection, operating schedule, or buffer zone distance.

Pivot with 1000 CFU/100ml at 787 gpm

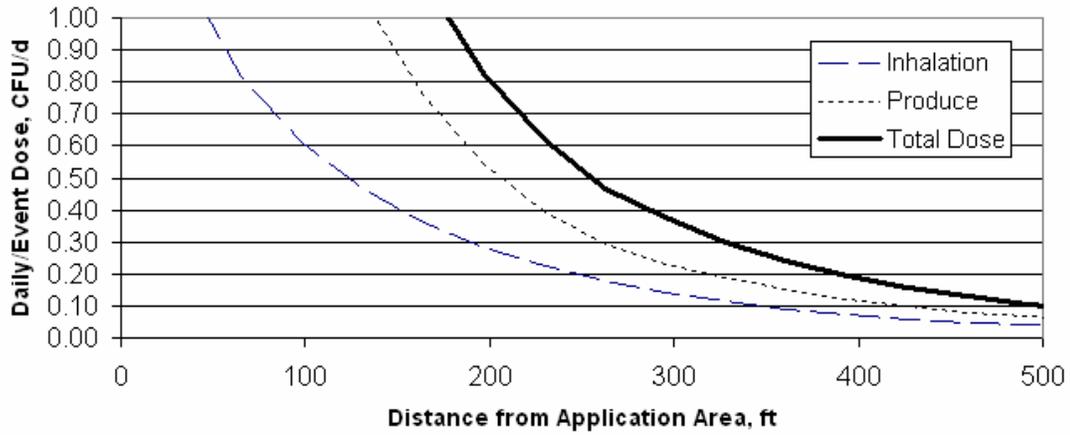


Figure 36. Dose contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml).

Pivot with 1000 CFU/100ml at 787 gpm

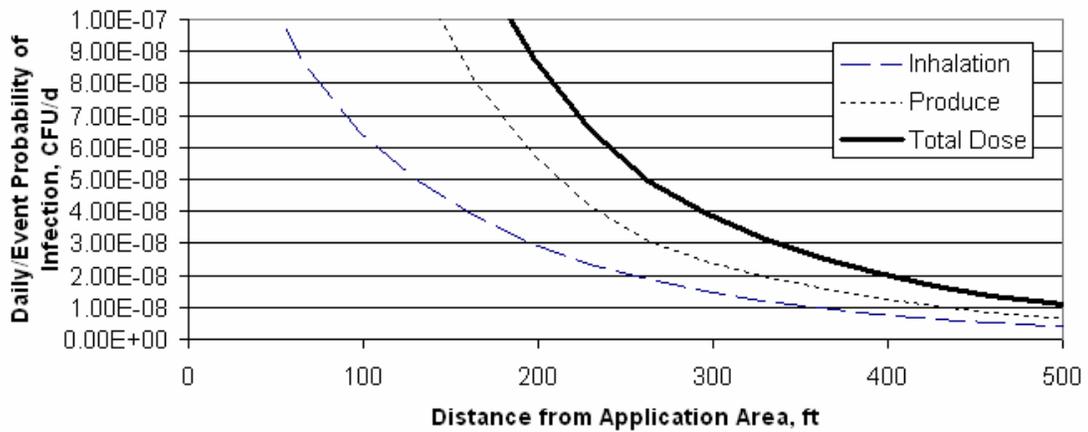


Figure 37. Daily risk contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml).

FEBRUARY 2006
 Pivot with 1000 CFU/100ml at 787 gpm

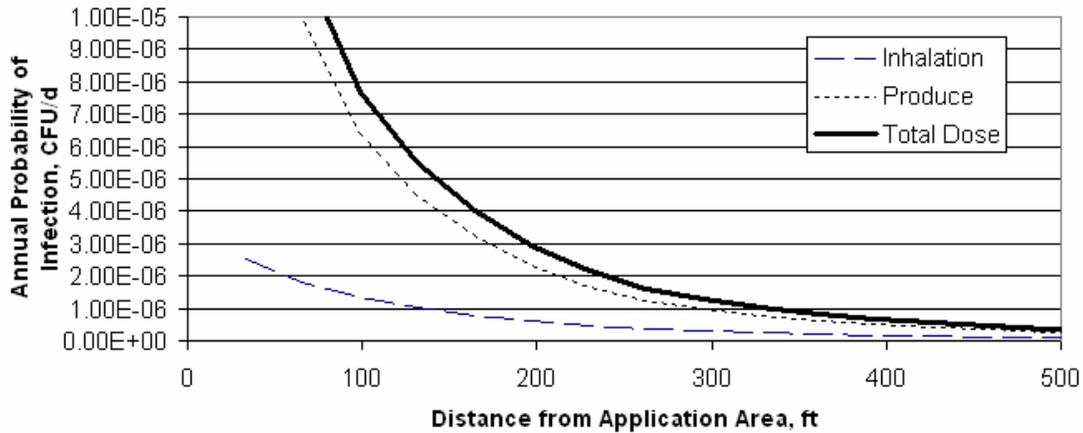


Figure 38. Annual risk contributions for *E. coli*: D stability, 2.5 mps at reduced wastewater loading (1,000 CFU/100 ml)

7.6.3 Results for Alternative Sprinkler Nozzles, 10^{-6} CFU/100ml

In reviewing the hypothetical design described in the first part of this section, it may be noted that the rotators and endgun are operated at high pressure and, as a result, have aerosolization and misting (< 0.2 mm) fractional efficiencies in the mid- to high end of the range available in Kincaid’s database (Table 5, page 65). The rotators, Kincaid Test No. 46 have a 0.1 mm fraction of 0.0027, while the end gun, Kincaid Test No. 17, has a 0.1 mm fraction of 0.0017.

The entries in Table 5 (page 65) are sorted in increasing order of 0.1 mm fractions. Thus, it is convenient to find alternative nozzle/pressure combinations with lower aerosolization and misting fractions. For example, the same end gun operated in Test No. 17 at 60 psi was also tested by Kincaid at 30 psi (Test No. 16). This test has a very low aerosolization fraction (0.1 mm fraction = 0.0003) and misting fractions (0.2 mm fraction = 0.001).

Further, a Wobbler-type ¼ inch nozzle operated at low pressure (Kincaid Test No. 37) also has very low aerosolization and misting efficiencies (also 0.0003 and 0.001 for 0.1 mm and 0.2 mm, respectively). If feasible, considering the water distribution and droplet kinetic energy considerations, these two alternative “low-E” nozzles could reduce the microbial transport and resulting risk significantly.

The MIRA analysis was re-run, using aerosolization and misting fractional efficiencies of 0.0003 and 0.001 for both the end gun and the rotators. Revised MIRA results are shown in Figure 39, Figure 40, and Figure 41 for these

alternative low aerosolizing efficiency (low E) sprinkler heads. The daily risk is now 5×10^{-6} at the 300 foot distance as seen in Figure 40 compared to 4×10^{-5} in Figure 27 with the same microbial loading (10^6 CFU/100ml) but the original high-E nozzles. The annual risk (Figure 41) is now between 1 and 2×10^{-4} at 300 feet - compared to about 1.2×10^{-3} in Figure 28 (page 83), and now very close to the EPA drinking water criterion of 10^{-4} discussed in Section 5.3, page 53.

Depending on the regulatory scenario and the level of infection risk that is considered acceptable, this analysis appears to show that the low aerosolization/misting nozzles (low-E nozzles), in combination with perhaps a one decade reduction in average wastewater loadings, will probably result in an acceptable design.

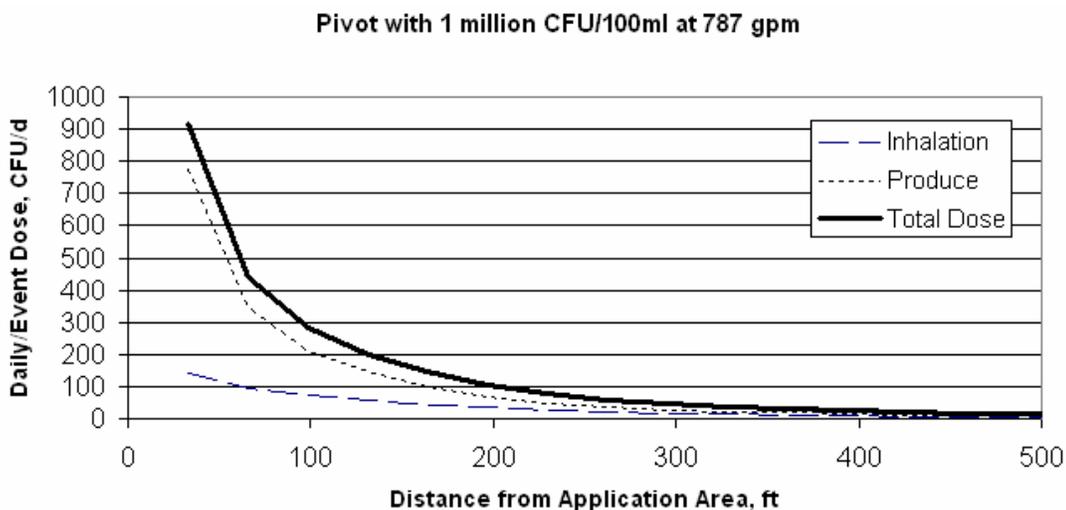


Figure 39. Dose contributions for *E. coli*: low-E nozzles at 10^6 CFU/100 ml, D stability, 2.5 mps.

Pivot with 1 million CFU/100ml at 787 gpm

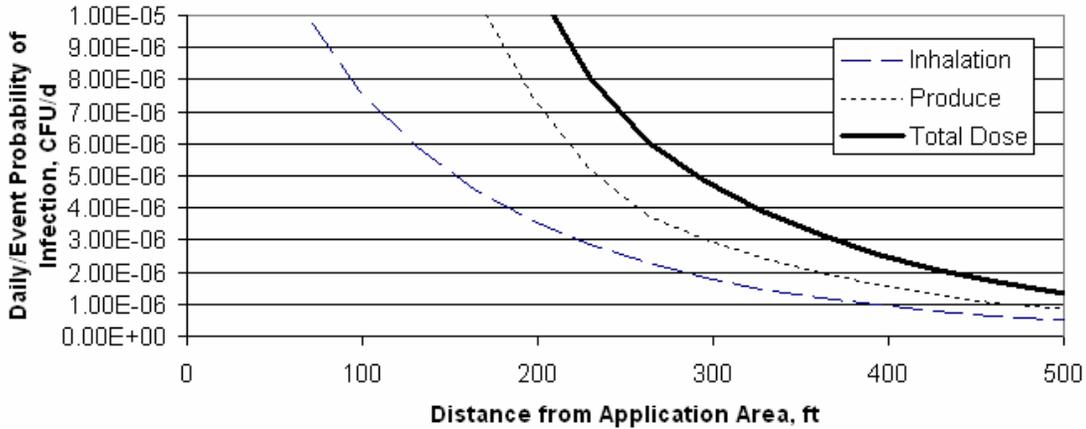


Figure 40. Daily risk contributions for *E. coli*: low-E nozzles at 10^6 CFU/100 ml, D stability, 2.5 mps.

Pivot with 1 million CFU/100ml at 787 gpm

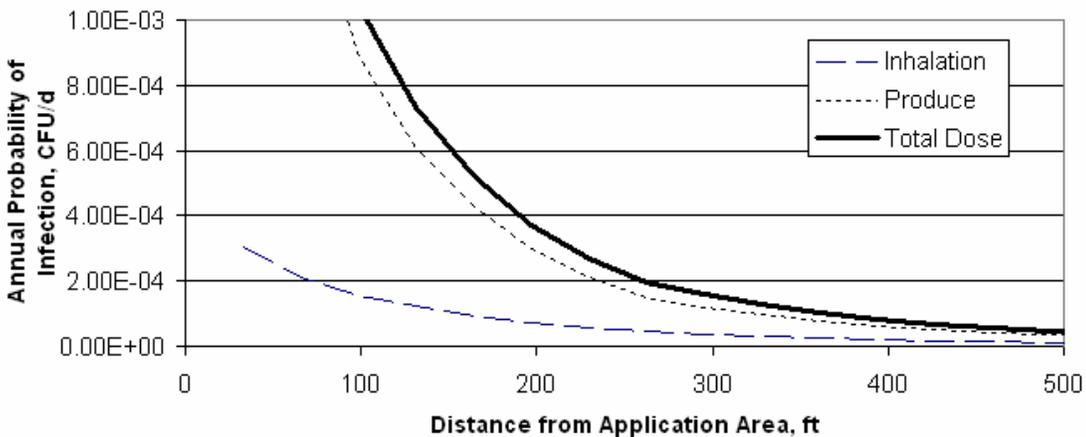


Figure 41. Annual risk contributions for *E. coli*: low-E nozzles at 10^6 CFU/100 ml, D stability, 2.5 mps.

7.7 Lessons Learned from Example Analysis

The hypothetical MIRA example discussed in this section is intended to demonstrate the nature of results that can be obtained during a wastewater land application design. The MIRA methodologies described above involve significant uncertainties, which must be acknowledged, similar to any chemical risk analysis. This analysis method may not be appropriate, in this stage of development, for wastewater permitting decision making, but it should be considered a risk reduction tool; a heuristic (i.e. exploratory problem

solving) tool for helping understand the how certain design decisions can be expected to reduce risk.

Among the concepts revealed by this hypothetical example are the following:

- *E. coli* loadings in wastewater that are less than 100 to 1,000 CFU/100 ml appear to represent very minimal risk at or beyond a typical 300 foot buffer zone.
- In addition to the aerosolized fraction, fine mist droplets (0.2 mm diameter) should also be considered in a risk analysis.
- Deposition of very fine mist on produce followed by ingestion can be a significant contributor to total risk of infection, particularly at higher wind speeds. Thus, simplified modeling without the deposition component, as proposed in the 1982 EPA guidance, may not be adequate under such conditions.
- The relative contributions of airborne versus deposition-based pathways are highly dependant on the dispersive conditions of the atmosphere.
- Many design parameters influence the risk estimates, and land application system designers should explore both microbial reduction and minimizing fine droplet formation, in addition to buffer zone restrictions, as methods to reduce risk.
- The MIRA results are thought to be far less uncertain when used in a relative sense (e.g. what is the relative change in risk due to reducing aerosolization efficiencies), rather than an absolute sense.

8. Conclusions and Recommendations

As discussed previously, air modeling and risk assessment methodologies are based upon previous and well established work. Modifications to these methodologies have been made to adapt them to particular circumstances surrounding wastewater land treatment operations. This enables site-specific determinations of 1) potential microbial risk, 2) modifications to irrigation systems and management, and 3) facility specific permit limits and conditions for regulatory purposes.

The exploratory MIRA simulations conducted to date, described in Sections 6 and 7, have already suggested a number of technical conclusions regarding microbial fate, transport, and risk analysis:

1. Fine droplets may contribute to microbial risk under high wind conditions, (previous work by EPA (1982) ignored the droplet fraction.)
2. Droplets larger than 0.2 millimeters (200 micrometers) do not transport significantly beyond the application area and may be neglected when analyzing risk at typical buffer zone distances.
3. Deposition of droplets and aerosol containing microbial pathogens on surfaces such as produce may be a significant pathway for exposure under windy conditions. Thus, if wastewater loadings are elevated, high-wind cut-off restrictions should be considered.
4. Worst-case conditions that lead to the greatest exposure and potential risk of infection are nighttime low-wind stable conditions, which maximize the inhalation pathway, and high-wind conditions, which maximize the deposition and produce ingestion pathway.
5. Microbial risk analyses methods suggested in this document incorporate many uncertainties. Uncertainties can be balance by conservative parameter selections such that the analyst can be assured that the true risk of infection are not likely to be greater than the risk estimated with this approach.
6. Regardless of absolute risk estimates, the methods described above may be most useful when used to explore the relative risk reduction that may be obtained by mitigation measures such as alternative nozzles and system pressures, treatment to reduce wastewater microbial levels, time of day and wind speed restrictions.

The following are recommendations for further work that would supplement and enhance the microbial risk assessment recommendations in this document for eventual implementation at permitted wastewater land treatment facilities.

1. *Microbial Wastewater Characterization.* The purpose of this activity would be to more rigorously characterize typical food processing, industrial, and municipal wastewaters in Idaho for wastewater-specific pathogenic microorganisms as well as for indicator species, as applicable. This information would be used as the microbial source term for drift/deposition modeling. The activity would provide microbial characterization at a point in time so that individual facilities do not need to conduct their own studies.
2. *Microbial Irrigation Water Characterization.* The purpose of this activity would be to characterize diverted surface irrigation waters in representative irrigation districts in Idaho for specific pathogenic microorganisms as well as for indicator species, as applicable. Modeling drift/deposition of microbial constituents from irrigated agriculture would provide an important reference point between common and longstanding irrigation practices, expected exposures, and wastewater land application practice. Such comparisons are important in making final decisions regarding risk and public health.
3. *Consideration of Future Development of Automated Irrigation Droplet Drift – Aerosol Modeling Tool:* The purpose of this activity is to continue the development of an integrated modeling tool utilizing an irrigation droplet drift model and aerosol dispersion models, in order to more accurately characterize microbial deposition and microbial densities along the flow path for both aerosols and the trajectories of larger droplets. This addition would involve hourly calculation of changing droplet size and more accurate transport and deposition modeling. Exposure to microorganisms in these various and transitioning states can be summed and utilized in microbial risk assessments. This work should not proceed until significant review and exercising of the current manual/spreadsheet model occurs.
4. *Field Calibration and Validation of Microbial Drift/Deposition Modeling.* The purpose of this activity would be to compare computer modeled microbial deposition / aerosolized microbial density results with field

measurements focused primarily on droplet/aerosol deposition, a previously understudied impact, which this study suggests may be important. For budgetary reasons, this activity should be limited to characterizing deposition onto deposition plates—an inexpensive analysis—to characterize both deposition and microbial densities indirectly. Depending upon the correlations obtained in field studies, modeling parameters can be modified and the model calibrated to actual Idaho field conditions.

5. *Development of Simplified Stand-Alone Bioaerosol Estimation Tools/ Tables.* Wastewater source term microbial characterization described in Item 1 above would be utilized in drift / deposition modeling. Modeling would be done on representative wastewaters utilizing typical irrigation systems and varying meteorological conditions. Microbial risk recommendations for site-specific scenarios would be pre-modeled and provided to the regulated community. This would include graphs and nomographs for facilities to consult to determine the appropriate irrigation system design, given site specific meteorological conditions and wastewater characteristics, to design and operate wastewater land treatment systems with low or no microbial risk without the need for time consuming and expensive site-specific modeling.
6. *Establish Bioaerosol Concentration and Deposition Target Levels.* Determine risk goals, and then generate backward risk analyses to develop conservative bioaerosol and deposition target levels that are protective. Such target levels, based on this work and other input, would be easier for the regulated community to use in their designs and for monitoring. This task would require significant programmatic dialogue concerning risk goals.
7. *Develop Step-by-Step Example Calculations.* This activity would involve development of an appendix to these recommendations, which takes the potential user of these recommendations through the calculations systematically, using an actual example, or series of examples, to further illustrate how the equations are used, which units are used, how units cancel, and how output from calculations are subsequently used as input into successive calculations.
8. *Apply these recommendations to Site-Specific Permitting Circumstances.* It is important to apply the methodologies described herein to actual regulatory circumstances in an internal, advisory mode. By doing this,

FEBRUARY 2006

other modifications and adaptations may suggest themselves for inclusion in revisions to these recommendations.

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Appendix A – Types of Idaho Wastewater Land Treatment Facilities

There are several different types of wastewater land treatment facilities in Idaho. These generate unique wastewaters that are briefly described below.

Potato Processing Wastewater

There are several dehydration potato processing facilities in Idaho. For these facilities, wastewater is a combination of wastewater from potato washing and fluming (silt wastewater) and potato processing (process wastewater). The silt wastewater is clarified to remove silt in a conventional clarifier, Delta Stak[®], settling pit, or other equivalent process. Potato process wastewater is generally a combination of cooker water, blancher water, general plant sewer water (non-septic), and other flows. This wastewater receives primary screening, after which it generally goes to a clarifier to remove suspended solids. Solids from clarifier underflow are removed with a vacuum filter drum, centrifuge, hydroclone, or equivalent process. Filtrate or concentrate from clarifier underflow is routed either back to the clarifier or on to land treatment, depending on how the system is plumbed. Both the silt and process wastewater is generally combined after clarification and conveyed to land treatment fields. None of the dehydration potato processing facilities in Idaho have long-term storage facilities, and most do not have short term storage either.

One of these facilities has advanced treatment/by-product recovery where high strength blancher and cooker wastewater is diverted and differentially treated through reverse osmosis, ultra-filtration, and evaporative processes to remove organic and inorganic constituents from the wastewater. The concentrated by-product is sold for various food uses.

There are three French fry potato processing facilities in Idaho. The wastewater treatment train for French fry processing is similar to that for dehydration potato processing, the exception being the need for fat, oil, and grease (FOG) removal in French fry processing. Only one facility has a dissolved air flotation unit process for FOG removal. The other two rely on clarifier skimming. Two of these facilities have storage to which post clarifier water is routed. One has an extensive system of natural ponds that allows for long detention times during which significant evaporation and facultative treatment takes place.

Sugar Beet Processing Wastewater

There are three sugar beet processing facilities in Idaho. These plants are rather complex with respect to wastewater generation and respective treatment processes, and there are significant differences among them in certain aspects of wastewater treatment. There are several wastewater streams generated by these facilities, each having a unique origin and characteristics. The flume system generates wastewater from flume transport of beets into the plant. The lime water system generates wastewater from the sugar purification process. The scrubber water system generates wastewater from pulp drying and coal-fired boiler scrubber systems. The fly ash system generates wastewater from slurring coal-fired boiler bottom ash to ash ponds. The condenser system generates condensate from the sugar crystallization process. And finally, a sanitary wastewater flow is generated from the plant, but does not generally mix with wastewater for land treatment. These wastewaters are typically stored in separate storage structures, but can be wasted to other storage structures at specific times during, and at the close of, campaign.

Wastewater quality from sugar beet processing varies considerably depending upon the types of processes taking place. The major processing phases at these sugar facilities are the beet slice campaign, in which beets are sliced and processed to sugar juice and sugar, and the juice run, which takes place after the beet slice and only processes sugar from sugar juice made and stored during the beet slice. These two processing phases generate different proportions and types of wastewater. Wastewater quality also depends upon the stage of the particular campaign and the age of the vegetative materials being processed. Biological degenerative processes affect older beets and thus they generate more waste during processing than fresh beets. Sources of coal for coal-fired boilers may influence the amount of sulfur in waste streams as well as the sulfur's isotopic signature. As mentioned above, wastewater from certain closed systems is wasted into wastewater storage ponds periodically and at the close of campaign. These events can significantly change wastewater quality over time. At two of these facilities, condensate wastewater generated is separated from other wastewater and is either re-used for boiler feed water or both re-used and land applied. A third facility does not isolate or re-use its condensate wastewater stream, nor does this facility land apply wastewater at this time.

These facilities have storage ponds that receive wastewater from several systems and are large enough to have significant detention times. Two of these

facilities are coal-powered, and the wastewater is influenced by sulfur from the coal. Sulfur can influence wastewater through the wasting of both scrubber pond wastewater and fly and bottom ash pond wastewater to wastewater storage ponds. It should be noted that the facilities obtain coal from more than one source, and coal from these sources is known to vary in its sulfur content. Condensate wastewater is stored and either re-used in processes or land applied at two facilities.

Municipal Wastewater

There are about ninety municipal wastewater treatment plants in Idaho which land apply wastewater. There are a number of different unit processes that may be utilized in the treatment of municipal wastewater. Wastewater from municipal collection systems typically undergoes primary treatment (solids removal through screening and primary clarification), then secondary (biological) treatment and clarification. After this, wastewater can be disinfected and discharged to surface water immediately or stored in lagoons for either discharge to surface water or land treatment (land application) at a later time.

Meat Processing Wastewater

There are three meat processing facilities in Idaho that land apply wastewater. One is a cattle slaughter facility, another a renderer, and a third is a food processor utilizing bulk prepared meats to manufacture meat-based food products. These employ various primary and secondary treatment prior to land application, and have variable amounts of storage.

Cheese Processing Wastewater

There are several cheese/whey processing plants in Idaho. The disposition of wastewater from these plants is varied. It is either discharged to a waste silo, trucked offsite, and immediately land applied; discharged to small anaerobic lagoons and then land applied; or discharged into highly aerated cells, then into large facultative lagoons with long detention times, then land applied.

Fertilizer Processing Wastewater

There is one fertilizer production facility in Idaho that land applies wastewater. Wastewater from this phosphorus plant undergoes primary treatment, and then pre-treatment (pH adjustment) if needed, before it enters a surge pond. After a

FEBRUARY 2006

moderately short detention time, wastewater either combines with municipal wastewater flows from a municipal facility or is routed to a wastewater land treatment site.

Appendix B – Typical Wastewater Land Treatment Systems

Wastewater land treatment systems are typically either overland flow, rapid infiltration, or slow rate systems. Overland flow systems are designed to apply wastewater at a few inches per week so that sheet flow occurs across the site. Wastewater interacts with soils, vegetation, and biological surface growths. Wastewater is collected at the end of the field and usually discharged to surface water, with appropriate surface discharge permits (Crites et al. 2000, page 14). There are no overland flow systems currently in Idaho.

There are several rapid infiltration systems permitted under the WLAP program in Idaho. These systems use flooding basins, which are dosed for denitrification of nitrogen, and COD and pathogen removal. The Wastewater-Land Application Permit Regulations, IDAPA 58.01.17.200.13, define rapid infiltration basins as percolating between 20 and 600 feet of wastewater per year, far more than either overland flow or slow rate systems. Permitted rapid infiltration systems in Idaho treat municipal wastewater streams exclusively.

Slow rate systems are the most common in Idaho. These systems are typically managed as agronomic operations in Idaho, with hydraulic and nutrient loadings in accordance with the evapotranspirative and nutrient needs of the crop. Forested areas are commonly used in northern Idaho. Areas having native vegetation are sometimes used.

Constituent and hydraulic loading rates vary among facilities. It has been the case that industrial, and in some cases, municipal, facilities have been grandfathered into the regulatory program practicing high loading rates, which either could have, or did, adversely impact the environment. Such facilities have been, and are, issued permits with compliance condition activities, which phase out high constituent loading and phase in appropriate loading rates, which rates are determined by scientific and engineering studies.

Just as facility types utilizing wastewater land treatment are varied in Idaho, so are irrigation management practices. During the growing season, most slow rate facilities practice sprinkle irrigation. Sprinkler irrigation systems utilized include pivot, linear move, wheel line, solid set, hand line, and big gun.

Pivot systems consist of one to several spans of pipe mounted on wheeled structures which rotate around one center point in a circle. Sprinklers are

mounted at intervals along the spans. Pivots can irrigate areas from a few acres to several hundred acres, and can be full, half, or partial circles. Sprinkler heads can be mounted on top of the span, or can be mounted on the ends of drop tubes to be closer to the ground.

Linear move irrigation systems are similar to pivots, except they travel in a straight line across a field rather than pivoting around a center point. Wheel lines consist of shorter spans of aluminum pipe, which are typically about waist high. Wheels are mounted at each end of the span, and are manually rolled forward to a different position after each irrigation set. Hand lines are similar to wheel lines, except that they have no wheels and must be moved by lifting and carrying rather than rolling.

Solid set systems consist of piping trenched and buried (or otherwise immobilized) with risers and sprinklers mounted along the pipe. These irrigation systems are permanent and non-movable installations.

Big guns are infrequently used for wastewater land application in Idaho during the growing season. They are high pressure-high discharge irrigation systems which can be either mobile or permanently installed. Uniformity of distribution is generally poor for big gun systems. A solid set system using big guns is shown in Figure 42.



Figure 42. Solid set big gun irrigation system at a potato processing wastewater land application site in Eastern Idaho.

Some facilities still practice furrow and flood irrigation instead of sprinkler irrigation during the growing season. Furrow irrigation is usually done with siphon tubes or gated pipe at the upper end of the field.

Irrigation systems described above can also be utilized for the non-growing season, but special winterizing procedures must be used to adapt sprinkler irrigation systems to winter conditions. Such modifications include installing of drag tubes (further described below), exchanging sprinkler heads with splash plates, and removing wastewater emitters adjacent to pivot wheels to prevent freezing, and insuring that rapid and adequate drainage of the irrigation system takes place, also to prevent freezing. Big gun and furrow/flood systems require little to no modification for non-growing season utilization.

FEBRUARY 2006

Appendix C – Wastewater Land Application Practices in Other Areas

Wastewater land treatment is practiced in one form or another in every state of the United States. It is beyond the scope of this appendix to describe respective programs of these states; suffice it to say that certain states have better developed programs, often due either to the particular need to reuse water in water poor areas, to address environmental issues caused in part by discharge to surface water, or to the magnitude of particular industries such as food processing, which present particular wastewater treatment needs. Wastewater land application is also practiced worldwide. In water scarce regions such as the Middle East, particularly Jordan and Saudi Arabia, national policies have been instituted to reuse all treated municipal effluent (FAO 1992; Section 1.1). Case studies of wastewater land treatment internationally (in Tunisia, India, Kuwait, Mexico, and elsewhere) are discussed in FAO (1992; Section 9).

There are many examples of successful slow rate systems in the United States. Crites et al. (2000; p. 11) describe several examples. The largest slow rate system is an 8,000 acre operation in Dalton, Georgia. Another slow rate system utilizing forest species is in Clayton County, Georgia, and has been in operation since 1981. It is often more expedient to utilize perennial grass crops on land treatment sites. They typically have longer growing seasons and higher water and nutrient uptake, and are agronomically less intensive. But a 5,000 acre land treatment operation in Muskegon, Michigan utilizes a majority of annual crops (corn, soybean) in addition to alfalfa.

Rapid infiltration is of benefit in arid regions (e.g., California, Arizona, Israel) serving both for municipal wastewater treatment as well as ground water recharge, aquifer storage, and subsequent reuse of treated water. There are few industrial wastewater rapid infiltration systems due to the high treatment needs of such wastewaters (Crites et al., 2000; p. 12, 313).

As discussed above, overland flow is utilized where soil permeabilities may be lower, and hydraulic loading may need to be high. A successful and large municipal overland flow system is in Davis, California. Industrial wastewaters such as those from a soup producer in Paris, Texas, and a tomato processor in Davis, California, have successfully utilized overland flow for wastewater land treatment (Crites et al., 2000; p 312-313).

FEBRUARY 2006

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Appendix D – Microorganisms in Municipal and Food Processing Wastewaters in Idaho

Table 7 and Table 8 below show limited data sets for total and fecal coliform content of Idaho municipal wastewaters permitted for land application. Data are from DEQ's database of wastewater land application program facility monitoring. Maximum, minimum, and median values for each monitoring station data set are given in addition to number of data points considered and standard deviation of the data sets. For both total and fecal coliform data, wide variations between maxima and minima in municipal wastewaters are evident. Data often show large standard deviations for the data sets indicating high variability in microbial content. Median levels of total and fecal coliform may also vary widely. Several factors influence these levels including whether the wastewater had been disinfected, the degree of disinfection, time elapsed since disinfection, strength of wastewater, and detention time of wastewater storage, among several other factors.

Food processing wastewater can have a considerable microbial content. Raw materials such as potatoes and sugar beets are harvested, washed, and processed. Soil adherent to raw food material has high microbial content, which is passed to wastewater during washing and fluming of the crop. In the case of cheese processing, pasteurized milk, which should have very low microbial content, is processed into cheese. During the course of cheese/whey processing, and subsequent transport, storage, and treatment of the resulting wastewater, a microbial load develops due to the presence of predisposing factors for microbial growth described above.

Table 7 and Table 8 also show limited data sets of total and fecal coliform data for food processing wastewaters in Idaho, including cheese, potato, and sugar beet processing. As noted above for municipal wastewater, both total and fecal coliform data show wide variations between maxima and minima, as well as large standard deviations. Note the substantial decrease in fecal coliforms between monitoring stations WW-000501, which is from the clarifier, and WW-000503, located at a pond which is about half way through a facultative pond treatment process. Total and fecal coliform levels in condensate wastewater from sugar beet processing (WW-005003) are significantly lower than other sugar beet processing wastewaters (WW-005001 and WW-004901).

This is likely because the level of organic material in condensate is much lower than in other wastewaters.

Table 7. Total coliform counts (CFU/100 mL) in various wastewaters in Idaho.

Total Coliform Sampling Data for Various Land Applied Wastewaters and Supplemental Irrigation Waters							
Facility Type	Sampling Station	Wastewater Sample Type	Maximum	Minimum	Median	Number of Analyses	Standard Deviation
Cheese Processing	WW-004201	Secondary Pond Effluent	160000	2400	81200	2	NA
Meat Processing	WW-005401	WW to Land Application	240000	9	305	12	69153
Meat Processing	WW-005501	WW from Aerobic Pond	240000	1000	2400	16	60527
Meat Processing	WW-005502	WW from Storage Lagoon	25000	10	170	15	6366
Municipal	MU-003701	WW to Field HR-1-A	2400	4	460	17	1146
Municipal	MU-003705	WW to Field HR-1-B	2400	3	23	34	787
Municipal	MU-011301	WW to Land Application	2200	1	29	11	777
Municipal	SW-011301	Two Mouth Creek Upstream	2000	1	45	25	543
Municipal	SW-011302	Two Mouth Creek Downstream	370	2	50	25	104
Municipal	SW-011303	Goose Creek Upstream	330	1	32	25	76
Municipal	SW-011304	Goose Creek Downstream	310	1	40	24	72
Municipal	WW-003701	WW to Land Application	2400	43	362	18	910
Municipal	WW-006701	WW to Land Application	20000	10	21	8	7551
Municipal	WW-006801	WW to Land Application	43	10	26.5	2	23
Municipal	WW-008601	WW from Sanitary Lagoon	1600	0.8	80	13	468
Municipal	WW-008602	WW from Lagoon Underdrain	9	0.01	0.1	11	3
Municipal	WW-008603	WW from Lagoon Underdrain	33	0.03	0.375	14	9
Municipal	WW-008801	Disinfected WW	45000	30	2000	7	16695
Municipal	WW-009601	WW to Land Application	20	4	10	7	6
Municipal	WW-010802	Mores Creek Downstream	2400	23	350	4	1094
Municipal	WW010901	WW to Land Application	22	4	13	2	13
Municipal	WW-011301	WW to Land Application	240	2	121	2	168
Municipal	WW-011502	WW to Rapid Infiltration Basin	44000	16	530	36	8464
Municipal	WW-014102	WW to Land Application	73	1	6.5	16	20
Municipal	WW-015301	WW to Rapid Infiltration Basin	66000	2600	17800	29	17962
Potato Processing	WW-000505	WW from Sanitary Lagoon	53000	9	275	22	13009
Sugar Beet Processing	WW-004901	WW to Land Application	16000000	1500	300000	69	3233212
Sugar Beet Processing	WW-004902	Irrigation Water	200000	1100	2400	10	62576
Sugar Beet Processing	WW-005001	WW to Land Application	35000	10	11000	8	13202
Sugar Beet Processing	WW-005003	Condensate WW	3800	2	10.5	8	1424

Notes:
 1) WW = wastewater
 2) Units are CFU/100 mL
 3) Many of the sample results are expressed as '>' (greater than) a certain value. These results are converted to numeric values to calculate summary values shown.

Table 8. Fecal coliform counts (CFU/100 mL) of various wastewaters in Idaho.

Fecal Coliform Sampling Data for Various Land Applied Wastewaters and Supplemental Irrigation Waters							
Facility Type	Sampling Station	Wastewater Sample Type	Maximum	Minimum	Median	Number of Analyses	Standard Deviation
Meat Processing	WW-005401	WW to Land Application	2400	75	350	8	1012
Meat Processing	WW-005501	WW from Aerobic Pond	240000	3	2400	16	59015
Meat Processing	WW-005502	WW from Storage Lagoon	25000	3	43	14	6663
Municipal	WW-007001	WW to Land Application	2400	2000	2200	2	283
Municipal	WW-008601	WW from Sanitary Lagoon	96	0.03	0.96	15	25
Municipal	WW-008602	WW from Lagoon Underdrain	7	0.01	0.07	3	4
Municipal	WW-008603	WW from Lagoon Underdrain	90	0.01	0.455	6	37
Municipal	WW-010802	Mores Ck Downstream	2400	4	232	4	1142
Municipal	WW-014102	WW to Land Application	18	1	1	17	6
Municipal	WW-015301	Influent to Rapid Infiltration	23000	167	4500	28	5351
Potato Processing	WW-000501	WW from Clarifier	20000000	100	660000	163	3982580
Potato Processing	WW-000503	WW from Pond #7	456	194	255.5	12	3826552
Potato Processing	WW-000505	WW from Sanitary Lagoon	3100	10	200	14	1031
Sugar Beet Processing	WW-004901	WW to Land Application	82000	300	2400	16	22368
Sugar Beet Processing	WW-004902	Irrigation Water	2400	23	2400	8	889
Sugar Beet Processing	WW-005001	WW to Land Application	10000	1	2400	9	2993
Sugar Beet Processing	WW-005002	WW before treatment	24000	10	2400	10	7421
Sugar Beet Processing	WW-005003	Condensate WW	1700	1	18	3	976

Notes:
 1) WW = wastewater
 2) Units are CFU/100 mL
 3) Many of the sample results are expressed as '>' (greater than) a certain value. These results are converted to numeric values to calculate summary values shown here.

One twelve-sample data set from an Idaho potato processor provides both fecal coliform and fecal streptococcus levels. The samples were taken from a

midpoint in a series of facultative ponds. Fecal coliform levels ranged from 194 to 456 colony forming units per 100 milliliters (CFU/100 mL), the median being 255 CFU/100 mL. Fecal streptococcus levels ranged from less than 1 to 25 CFU/100 mL, the median being 9 CFU/100 mL. Fecal coliform to fecal streptococcus ratios ranged from 15:1 to 395:1, the median being 39:1.

Microbial content of cheese processing wastewaters have been the focus of recent investigation in Idaho. Preliminary data representing a very limited data set for a variety of pathogenic organisms which have potential to occur in these wastewaters are shown in Table 9. Further investigation is needed to characterize microbial content of cheese processing wastewaters.

Table 9. Microbial content of cheese processing wastewaters in Idaho.

Other Microbial Sampling Data for Various Land Applied Wastewaters				
<u>Species</u>	<u>Maximum</u>	<u>Minimum</u>	<u>Median</u>	<u>Number of Analyses</u>
WW-010301 - Cheese Processing WW				
Fecal Streptococcus	5800000	2400	15000	11
Pseudomonas Aeruginosa	16000	4	300	11
Salmonella	ND	ND	ND	11
Listeria Monocytogenes	D	ND	ND	11
WW-004201 - Cheese Processing WW				
E. Coli O157:H7	ND	ND	ND	2
Salmonella	ND	ND	ND	2
Staphylococci Aureus	140	ND	ND	2
Listeria Monocytogenes	ND	ND	ND	2
Campylobacter	D	ND	--	2

Notes:
 1) WW = wastewater
 2) ND = no detect
 3) D = detect
 4) Units are CFU/100 mL
 5) Many of the sample results are expressed as '>' (greater than) a certain value. These results are converted to numeric values to calculate summary values shown here.

FEBRUARY 2006

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Index

- (AERMIC) Dispersion Model
 - AERMOD, 28
- acute exposure*, 44
- AERMOD, 28, 31
- aeromicrobiology, 7
- aerosol drift, xiii, 3, 5, 9
- aerosol size category
 - treatment in spreadsheet, 66
- aerosol source strength, 19
- aerosolization, v, 3, 7, 12, 99
- aerosolization efficiency, 11, 19, 20
- algorithms, 28, 31
- alveolar region, 49
- Amdur, 48, 97
- American Meteorological Society/Environmental Protection Agency Regulatory Model Improvement Committee* (AERMIC), 28
- annual risk
 - example, 82
- atmospheric conditions
 - typical, 81
- atmospheric oxidants, 23
- atmospheric stability, 31
- Bad Bug Book, 38
- Baes, 50, 51, 97
- bioaerosol stability, 13
- bioaerosols, 7, 12, 13, 14, 73
- box model, 27
- bronchi, 48
- bronchioles, 48
- Brooks, 12, 13, 97
- buffers
 - vegetative, 6
- calibration
 - aerosol models, 71
- Campylobacter jejuni*, 38
- CFU content, 19, 20
 - aerosol, 18
- characteristics
 - wastewater facilities, 2
- cheese, 2, 103, 111, 113
- cheese processing wastewater, 9
- cheese/whey processing plants, 103
- chemical carcinogens, 54
- chemicals of potential concern* (COPCs), 43
- chronic exposure*, 44
- ciliated cells, 49
- Clean Water Act, 1
- Colony Forming Units (CFU), 14, 75
- concentration
 - downwind, 17, 18
- condensate wastewater, 111
- condensate wastewater stream, 102
- constituent release types
 - aerosols, 25
- constituents, xiii, 3
- coordinate system
 - source, 30
- Cox, 13, 97
- Crites, 1, 97, 105, 109
- Crump, 44, 97
- culturability*, 12, 99
- daily risk
 - example, 82
- Davis, California, 109
- dehydration potato processing, 101
- Delta Stak[®], 101
- DEM files, 31
- denitrification, 105
- desiccation, 13, 23
- die-off
 - factors affecting, 22
- die-off factor, 22
- die-off kinetics, 13
- die-off rates, 12, 13
- diffusion, 7, 24, 25, 49
- Digital Elevation Model (DEM), 31
- dispersion model, 18
 - aerosols, 24
- DNA, 13

- dose-response analysis, 35
Dose-response assessment, 35
 drag tube, 5
 drag tubes, 4, 5, 107
 drift, v, xiii, xv, 3, 4, 5, 6, 9, 72, 94, 95
 drop tubes, 5, 106
 duration
 land application, 51
E. coli, 12, 13, 15, 36, 38, 49, 55, 99
 Echovirus, 15, 40
 emission-to-concentration conversion
 factor, 32
 end gun
 modeling of, 76
 enteric microorganisms, 9
 Enterotoxigenic *Escherichia coli*, 38
 enterovirus
 die-off rate, 12
 epithelium lining, 49
Escherichia coli, 40, 55, 97, 99
 E. coli, 12
event duration
 (ED), 49
Exposure Factors Handbook, 50
 EPA, 52
 exposure threshold, 36
 farmers, 52
 fat, oil, and grease
 (FOG), 101
 fecal coliform, 9, 12, 13, 14, 15, 36,
 111, 112
 aerosol, 23
 fecal contamination, 35
 fecal streptococci
 die-off rate, 12
 fecal streptococci,, 14, 15
 fecal streptococcus ratios, 113
 fertilizer production facility, 103
 fomites, 46
 frequency
 land application, 51
 furrow and flood irrigation, 107
 Gaussian diffusion equation, 25
 Gaussian steady-state models
 aerosols, 25
 geometric die-off, 12
 Gerba, 9, 40, 97, 98, 99
 gram-positive, 12
 gram-staining process, 12
 Guidelines for Land Application of
 Municipal and Industrial
 Wastewater, 3, 97
 Haas, 37, 38, 40, 41, 98, 99
 Handbook for Land Application of
 Municipal and Industrial
 Wastewater, 3, 97
 Havelaar, 40, 100
hazard quotient, 45
 health, v, xiii, 1, 2, 3, 35, 36, 37, 38,
 46, 56, 57, 59, 75, 94
 Heidelberg, 13, 99
 hepatic microsomal mixed-function
 oxidase, 45
 high wind speed, 67
 Hofman, 99
 homegrown produce, 47
 hydroclone, 101
hypothesis of cooperative interaction,
 39
hypothesis of independent action, 39
 Idaho Department of Environmental
 Quality, xiii
 (DEQ), xiii, 1
 illness, 35, 45, 46
immune competence
 acquired immunity, 44
 India, 109
 Industrial Source Complex Model
 (ISC3), 59
Industrial Source Complex Short Term
 model
 (ISCST3), 28
 infection, xiv, 17, 35, 36, 37, 38, 39,
 44, 45, 46, 47, 48, 52, 53, 54, 71, 82,
 83, 87, 92
 from land application, xiv, 47
 probability, 52
infective dose, 37, 44, 55
 infectivity model, 37
 infectivity testing, 44
 ingestion exposure, 48
 inhalation rate (*IR*), 49

- inhaled dose, 49, 54
- Initial Dispersion Coefficients
aerosols, 29
- inorganic salts, xiii, 3
- irrigation systems
types, 105
- ISCST3, 28, 31, 32
- Johnson, 14, 99
- Jordan, 109
- Katzenelson, 14, 15, 73, 99, 100
- Kincaid, 21, 59, 60, 61, 63, 67, 76, 89, 99
- Kincaid, D., 60
- Kuwait, 109
- land application facilities
currently permitted, xiii, 1
- land application permit program, 2
- large particles, 48
- Linear move systems, 5
- linearized multistage procedure, 44
- Lubbock, Texas, 15
- macrophages, 49
- Maier, 7, 14, 98, 99
- material application rate, 20
- maxima
fecal coliform, 111
- McEwen, 14, 99
- median infective dose*
(ID50), 35
- median viability decay rates, 12
- meteorological parameters
ISC3, 31
- Mexico, 109
- microbial constituents, xiii, 1
- microbial risk assessment*
(MIRA), xiii, 1, 43
- Middle East, 109
- minima
fecal coliform, 111
- mist droplets
wind driven, 69
- modeling
aerosol, 17
- Monte Carlo*
techniques, 47
- mucociliary elevator, 49
- mucociliary escalator, 76
- mucus-coated
surfaces, 49
- municipal wastewater, xiii, 1, 2, 14, 15, 97, 99, 103, 104, 105, 109, 111
- Muskegon, Michigan, 14, 109
- mycobacteria, 14, 15
- nasopharyngeal region, 76
- nasopharyngeal surfaces, 49
- Nasser, 13, 99
- National Academy of Sciences, 99
- National Weather Service*, 32
(NWS), 31
- Nelson
sprinkler, 64
- nomographs, 72, 95
- open air factors*
(OAF), 7
- organic material, xiii, 3, 112
- organism die-off
aerosol, 19
- outputs
aerosol modeling, 32
- overland flow, 105, 109
- oxygen toxicity, 7
- Parker, 14, 73, 99
- pathogen content, 19
- pathogen removal, 105
- pathogenic microorganisms, xiv, xv, 9, 35, 44, 46, 55, 94
- pathogenic organisms, 9, 35, 113
- pathogens, 12
- persistence factors, 31, 32
- Pillai, 14, 99
- pivot arm rotators
dispersion modeling of, 76
- Pleasanton, California, 14
- Poisson, 39, 40, 48, 52, 55, 100
- Poliovirus III, 40
- polymerase chain reaction (PCR), 13
- Poon, 12, 99
- potato processing wastewater, ii, 4, 5, 6, 14, 73
- potato washing, 101
- probability of infection, 38
example, 82

- produce
 - deposition on, 48
 - ingestion of, 80
 - pathogen concentration, 50
- Rafters, George, 1
- Rainbird
 - sprinkler, 64
- rapid infiltration systems, 105, 109
- raw food material, 111
- receptor distances, 30, 69, 71
- reference dose*, 37, 55
- Regli, 37, 39, 40, 98, 99
- relative humidity, 7, 13, 14, 15, 22, 34, 62, 71, 97
- Release Height
 - aerosol emissions, 29
- renderer, 103
- respiratory tract, 48
- risk, v, xiii, xiv, 1, 13, 35, 36, 37, 40, 43, 44, 45, 46, 47, 48, 52, 53, 54, 55, 56, 72, 93, 94, 95, 97, 98, 99
- risk assessment, v, xiii, xiv, 1, 35, 37, 43, 45, 46, 56, 85
- risk criterion
 - EPA, 53
- Rotavirus, 40
- safety, xiii, 1, 2, 3, 37, 56
- Salmonella*, 38, 40, 52, 55
- sampling interval*, 20
- sampling plan
 - considerations, 19
- Saudi Arabia, 109
- schools, xiii, 3
- SCREEN3, 28, 30, 31, 32
- screening-level modeling
 - aerosols, 30
- Senniger
 - sprinkler, 64
- sensitive subpopulations, 35
- Sharp, 50, 51, 97
- Shigella* spp., 38
- Shor, 50, 51, 97
- silt wastewater, 101
- Sjoreen, 50, 51, 97
- Smith, 9, 98, 99
- solar radiation, 22
- solids removal, 103
- Sorber, 12, 14, 97, 99, 100
- source characteristics
 - for aerosol modeling, 17
- Source Location
 - aerosols, 30
- source strength
 - aerosol modeling, 18
- spatial variability
 - sampling, 20
- Spendlove, 6, 99, 100
- Stetzenbach, 14, 100
- Streptococcus* spp, 38
- subchronic exposure*, 44
- substrate, 9
- sugar beet, 2, 102, 111
 - facilities, 102
- Sugar Beet Processing Wastewater, 102
- surface deposition, 47
- Surface Treatment Rule*
 - EPA, 53
- surge pond, 103
- survival
 - parameters, 12
- Tchobanoglous, 9, 100
- Teltsch, 13, 14, 15, 73, 99, 100
- temperature
 - aerosol modeling, 31
- terrain height
 - aerosol modeling, 30
- Teunis, 40, 100
- total coliform
 - aerosol, 23
- total dose
 - example, 82
- toxic compounds, 22
- toys
 - deposition on, 47
- trachea, 48
- treatment
 - goal, 53
- Tunisia, 109
- ultraviolet light, 23
- Universal Transverse Mercator* (UTM), 30

Vegetative buffers, 6
viability
 parameters, 12
viability reduction factor, 22
virtual point sources
 aerosol emissions from sprinklers,
 29
volumes
 wastewater, 2
wastewater characterization, 19
*Wastewater-Land Application Permit
Regulations*
 (IDAPA 58.01.17), 2
wind direction, 31
wind drift, xiii
wind speed, 31
Wobbler, 60, 89
worst case
 conditions for, 77