Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality

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Waste from agricultural livestock operations has been a long-standing concern with respect to contamination of water resources, particularly in terms of nutrient pollution. However, the recent growth of concentrated animal feeding operations (CAFOs) presents a greater risk to water quality because of both the increased volume of waste and to contaminants that may be present (e.g., antibiotics and other veterinary drugs) that may have both environmental and public health importance. Based on available data, generally accepted livestock waste management practices do not adequately or effectively protect water resources from contamination with excessive nutrients, microbial pathogens, and pharmaceuticals present in the waste. Impacts on surface water sources and wildlife have been documented in many agricultural areas in the United States. Potential impacts on human and environmental health from long-term inadvertent exposure to water contaminated with pharmaceuticals and other compounds are a growing public concern. This workgroup, which is part of the Conference on Environmental Health Impacts of Concentrated Animal Feeding Operations: Anticipating Hazards—Searching for Solutions, identified needs for rigorous ecosystem monitoring in the vicinity of CAFOs and for improved characterization of major toxicants affecting the environment and human health. Last, there is a need to promote and enforce best practices to minimize inputs of nutrients and toxicants from CAFOs into freshwater and marine ecosystems. *Key words:* ecology, human health, poultry, swine, water contaminants, wildlife. *Environ Health Perspect* 115:308–312 (2007). doi:10.1289/ehp.8839 available via *http://dx.doi.org/* [Online 14 November 2006]

Background and Recent Developments

Concentrated animal feed operations and water quality. Animal cultivation in the United States produces 133 million tons of manure per year (on a dry weight basis) representing 13-fold more solid waste than human sanitary waste production [U.S. Environmental Protection Agency (U.S. EPA) 1998]. Since the 1950s (poultry) and the 1970s–1980s (cattle, swine), most animals are now produced for human consumption in concentrated animal feeding operations (CAFOs). In these industrialized operations, the animals are held throughout their lives at high densities in indoor stalls until they are transported to processing plants for slaughter. There is substantial documentation of major, ongoing impacts on aquatic resources from CAFOs, but many gaps in understanding remain.

Contaminants detected in waste and risk of water contamination. Contaminants from animal wastes can enter the environment through pathways such as through leakage from poorly constructed manure lagoons, or during major precipitation events resulting in either overflow of lagoons and runoff from recent applications of waste to farm fields, or atmospheric deposition followed by dry or wet fallout (Aneja 2003). The magnitude and direction of transport depend on factors such as soil properties, contaminant properties, hydraulic loading characteristics, and crop management practices (Huddleston 1996). Many contaminants are present in livestock wastes, including nutrients (Jongbloed and Lenis 1998), pathogens (Gerba and Smith 2005; Schets et al. 2005), veterinary pharmaceuticals (Boxall et al. 2003; Campagnolo et al. 2002; Meyer 2004), heavy metals [especially zinc and copper; e.g., Barker and Zublena (1995); University of Iowa and Iowa State Study Group (2002)], and naturally excreted hormones (Hanselman et al. 2003; Raman et al. 2004). Antibiotics are used extensively not only to treat or prevent microbial infection in animals (Kummerer 2004), but are also commonly used to promote more rapid growth in livestock (Cromwell 2002; Gaskins et al. 2002; Liu et al. 2005). In addition, pesticides such as dithiocarbamates are applied to sprayfields (Extension Toxicology Network 2003). Although anaerobic digestion of wastes in surface storage lagoons can effectively reduce or destroy many pathogens, substantial remaining densities of microbial pathogens in waste spills and seepage can contaminate receiving surface- and groundwaters (e.g., Burkholder et al. 1997; Mallin 2000). Pharmaceuticals can remain present as parent compounds or degradates in manure and leachates even during prolonged storage. Improper disposal of animal carcasses and abandoned livestock facilities can also contribute to water quality problems. Siting of livestock operations in areas prone to flooding or where there is a shallow water table increases the potential for environmental contamination.

The nutrient content of the wastes can be a desirable factor for land application as fertilizer for row crops, but overapplication of livestock wastes can overload soils with both macronutrients such as nitrogen (N) and phosphorous (P), and heavy metals added to feed as micronutrients (e.g., Barker and Zublena 1995). Overapplication of animal wastes or application of animal wastes to saturated soils can also cause contaminants to move into receiving waters through runoff and to leach through permeable soils to vulnerable aquifers. Importantly, this may happen even at recommended application rates. As examples, Westerman et al. (1995) found 3–6 mg nitrate (NO_3)/L in surface runoff from sprayfields that received swine effluent at recommended rates; Stone et al. (1995) measured 6–8 mg total inorganic N/L and 0.7–1.3 mg P/L in a stream adjacent to swine effluent sprayfields. Evans et al. (1984) reported 7–30 mg NO_3/L in subsurface flow draining a sprayfield for swine wastes, applied at recommended rates. Ham and DeSutter (2000) described export rates of up to 0.52 kg ammonium m^{-2} $year^{-1}$ from lagoon seepage; Huffman and Westerman (1995) reported that groundwater near swine waste lagoons averaged 143 mg inorganic N/L, and estimated export rates at 4.5 kg inorganic N/day. Thus, nutrient losses into receiving waters can be excessive relative to levels (~ 100–200 µg inorganic N or P/L)

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known to support noxious algal blooms (Mallin 2000). In addition to contaminant chemical properties, soil properties and climatic conditions can affect transport of contaminants. For example, sandy, well-drained soils are most vulnerable to transport of nutrients to underlying groundwater (Mueller et al. 1995). Nutrients can also readily move through soils under wet conditions (McGechan et al. 2005).

Presence of contaminants in water sources. The presence of many contaminants from livestock waste has been documented in both surface water and groundwater supplies in agricultural areas within the United States (e.g., Campagnolo et al. 2002; Kolpin et al. 2002; Meyer 2004). Urban wastewater streams also contain these contaminants, and efforts to accurately determine sources of contamination are under way (Barnes et al. 2004; Cordy et al. 2004; Kolpin DW, unpublished data). The U.S. Geological Survey (USGS) began pilot surveillance programs for organic wastewater contaminants in 1999 and expanded that effort to a national scale over the past 5 years (Kolpin et al. 2002). Recent USGS efforts have focused specifically on water quality in agricultural locations (Kolpin DW, unpublished data). Nutrient levels have been detected in high parts per million (milligrams per liter) levels; pharmaceuticals and other compounds are generally measured in low levels (ppb [micrograms per liter]). In Europe, surveillance efforts conducted in Germany documented the presence of veterinary pharmaceuticals in water resources (Hirsch et al. 1999).

Animal wastes are also rich in organics and high in biochemical oxygen-demanding materials (BOD); for example, treated human sewage contains 20–60 mg BOD/L, raw sewage contains 300–400 mg BOD/L, and swine waste slurry contains 20,000–30,000 mg BOD/L (Webb and Archer 1994). Animal wastes also carry parasites, viruses, and bacteria as high as 1 billion/g (U.S. EPA 1998). Swine wastes contain > 100 microbial pathogens that can cause human illness and disease [see review in Burkholder et al. (1997)]. About one-third of the antibiotics used in the United States each year is routinely added to animal feed to increase growth (Mellon et al. 2001). This practice is promoting increased antibiotic resistance among the microbial populations present and, potentially, increased resistance of naturally occurring pathogens in surface waters that receive a portion of the wastes.

Contaminant impacts. Some contaminants pose risks for adverse health impacts in wildlife or humans. The effects of numerous waterborne pathogens on humans are well known, although little is known about potential impacts of such microorganisms on aquatic life. With respect to nutrients, excessive phosphorus levels can contribute to algal blooms and cyanobacterial growth in surface waters used for recreation and as sources of drinking water. Research is beginning to investigate the environmental effects, including endocrine disruption and antibiotic resistance issues (Burnison et al. 2003; Delepee et al. 2004; Fernandez et al. 2004; Halling-Sorensen et al. 2003; Sengelov et al. 2003; Soto et al. 2004; Wollenberger et al. 2000). However, knowledge is limited in several crucial areas. These areas include information on metabolites or environmental degradates of some parent compounds; the environmental persistence, fate, and transport and toxicity of metabolites or degradates (Boxall et al. 2004); the potential synergistic effects of various mixtures of contaminants on target organisms (Sumpter and Johnson 2005); and the potential transport and effects from natural and synthetic hormones (Hanselman et al. 2003; Soto et al. 2004). Further, limited monitoring has been conducted of ecosystem health in proximity to CAFOs, including monitoring the effects on habitats from lagoon spills during catastrophic flooding (Burkholder et al. 1997; Mallin et al. 1997; Mallin et al. 2000).

Ecologic and wildlife impacts. Anoxic conditions and extremely high concentrations of ammonium, total phosphorus, suspended solids, and fecal coliform bacteria throughout the water column for approximately 30 km downstream from the point of entry have been documented as impacts of waste effluent spills from CAFOs (Burkholder et al. 1997; Mallin et al. 2000). Pathogenic microorganisms such as *Clostridium perfringens* have been documented at high densities in receiving surface waters following CAFO waste spills (Burkholder et al. 1997). These degraded conditions, especially the associated hypoxia/anoxia and high ammonia, have caused major kills of freshwater fish of all species in the affected areas, from minnows and gar to largemouth bass, and estuarine fish, including striped bass and flounder (Burkholder et al. 1997). Waste effluent spills also stimulated blooms of toxic and noxious algae. In freshwaters, these blooms include toxic and noxious cyanobacteria while in estuaries, harmful haptophytes and toxic dinoflagellates arise. Most states monitor only water-column fecal coliform densities to assess whether waterways are safe for human contact. World Health Organization (WHO) guidelines for cyanobacteria in recreational water are 20,000 cyanobacterial cells/mL, which indicates low probability of adverse health effects, and 100,000 cyanobacterial cells/mL, which indicates moderate probability of adverse health effects (WHO 2003). Yet fecal bacteria and other pathogenic microorganisms typically settle out to the sediments where they can thrive at high densities for weeks to months following CAFO waste effluent spills (Burkholder et al. 1997).

The impacts from CAFO pollutant loadings to direct runoff are more substantial after such major effluent spills or when CAFOs are flooded and in direct contact with surface waters (Wing et al. 2002). Although the acute impacts are often clearly visible—dead fish floating on the water surface, or algal overgrowth and rotting biomass—the chronic, insidious, long-term impacts of commonly accepted practices of CAFO waste management on receiving aquatic ecosystems are also significant (U.S. EPA 1998). One purpose of manure storage basins is to reduce the N content of the manure through volatilization of ammonia and other N-containing molecules. Many studies have shown, for example, that high nutrient concentrations (e.g., ammonia from swine CAFOs, or ammonia oxidized to NO_3, or phosphorus from poultry CAFOs) commonly move off-site to contaminate the overlying air and/or adjacent surface and subsurface waters (Aneja et al. 2003; Evans et al. 1984; Sharpe and Harper 1997; Sharpley and Moyer 2000; Stone et al. 1995; U.S. EPA 1998; Webb and Archer 1994; Westerman et al. 1995; Zahn et al. 1997). Inorganic N forms are added to the atmosphere during spray practices, and both ammonia and phosphate can also adsorb to fine particles (dust) that can be airborne. The atmospheric depositions are noteworthy, considering that a significant proportion of the total ammonium from uncovered swine effluent lagoons and effluent spraying (an accepted practice in some states) reenters surface waters as local precipitation or through dry fallout (Aneja et al. 2003; U.S. EPA 1998, 2000). The contributed nutrient concentrations from the effluent greatly exceed the minimal levels that have been shown to promote noxious algal blooms (Mallin 2000) and depress the growth of desirable aquatic habitat species (Burkholder et al. 1992). The resulting chronically degraded conditions of nutrient overenrichment, while not as extreme as during a major waste spill, stimulate algal blooms and long-term shifts in phytoplankton community structure from desirable species (e.g., diatoms) to noxious species.

A summary of the findings from a national workshop on environmental impacts of CAFOs a decade ago stated that there was “a surprising lack of information about environmental impacts of CAFOs to adjacent lands and receiving waters” (Thu K, Donham K, unpublished data). Although the knowledge base has expanded since that time, especially regarding adverse effects of inorganic N and P overenrichment and anoxia, impacts of many CAFO pollutants on receiving aquatic ecosystems remain poorly understood. As examples, there is poor understanding of the impacts of fecal bacteria and other microbial pathogens from CAFO waste effluent contamination on aquatic communities; impacts of antibiotic-resistant bacteria created from CAFO wastes on aquatic life; impacts of organic nutrient forms preferred by certain noxious plankton; impacts from the contributed pesticides and heavy metals; and impacts from these pollutants acting in concert, additively or synergistically. This lack of information represents a critical gap in our present ability to assess the full extent of CAFO impacts on aquatic natural resources.

Despite their widespread use, antibiotics have only recently received attention as environmental contaminants. Most antibiotics are designed to be quickly excreted from the treated organism. Thus, it is not surprising that antibiotics are commonly found in human and animal waste (Christian et al. 2003; Dietze et al. 2005; Glassmeyer et al. 2005; Meyer 2004) and in water resources affected by sources of waste (Glassmeyer et al. 2005; Kolpin et al. 2002). Although some research has been conducted on the environmental effects from antibiotics (e.g., Brain et al. 2005; Jensen et al. 2003), much is yet to be understood pertaining to long-term exposures to low levels of antibiotics (both individually and as part of complex mixtures of organic contaminants in the environment). The greatest risks appear to be related to antibiotic resistance (Khachatourians 1998; Kummerer 2004) and natural ecosystem functions such as soil microbial activity and bacterial denitrification (Costanzo et al. 2005; Thiele-Bruhn and Beck 2005).

Human health impacts. Exposure to waterborne contaminants can result from both recreational use of affected surface water and from ingestion of drinking water derived from either contaminated surface water or groundwater. High-risk populations are generally the very young, the elderly, pregnant women, and immunocompromised individuals. Recreational exposures and illnesses include accidental ingestion of contaminated water that may result in diarrhea or other gastrointestinal tract distress from waterborne pathogens, and dermal contact during swimming that may cause skin, eye, or ear infections. Drinking water exposures to pathogens could occur in vulnerable private wells; under normal circumstances community water utilities disinfect water sufficiently before distribution to customers. Cyanobacteria (blue–green algae) in surface water can produce toxins (e.g., microcystins) that are known neurotoxins and hepatotoxins. Acute and chronic health impacts from these toxins can occur from exposures to both raw water and treated water (Carmichael et al. 2001; Rao et al. 2002). Removal of cyanotoxins during drinking water treatment is a high priority for the drinking water industry (Hitzfield et al. 2000; Rapala et al. 2002). The WHO has set a provisional drinking water guideline of 1 µg microcystin-LR/L (Chorus and Bartram 1999). While there are no drinking water standards in the United States for cyanobacteria, they are on the U.S. EPA Unregulated Contaminant Monitoring Rule List 3 (U.S. EPA 2006).

Exposure to chemical contaminants can occur in both private wells and community water supplies, and may present health risks. High nitrate levels in water used in mixing infant formula have been associated with risk for methemoglobinemia (blue-baby syndrome) in infants under 6 months of age, although other health factors such as diarrhea and respiratory disease have also been implicated (Ward et al. 2005). The U.S. EPA drinking water standard of 10 mg/L NO_3–N and the WHO guideline of 11 mg/L NO_3–N were set because of concerns about methemoglobinemia. (Note: "nitrate" refers to nitrate–nitrogen). Epidemiologic studies of noncancer health outcomes and high nitrate levels in drinking water have reported an increased risk of hyperthyroidism (Seffner 1995) from long-term exposure to levels between 11–61 mg/L (Tajtakova et al. 2006). Drinking water nitrate at levels < 10 mg/L has been associated with insulin-dependent diabetes (IDDM; Kostraba et al. 1992), whereas other studies have shown an association with IDDM at nitrate levels > 15 mg/L (Parslow et al. 1997) and > 25 mg/L (van Maanen et al. 2000). Increased risks for adverse reproductive outcomes, including central nervous system malformations (Arbuckle et al. 1988) and neural tube defects (Brender et al. 2004; Croen et al. 2001), have been reported for drinking water nitrate levels < 10 mg/L.

Anecdotal reports of reproductive effects of nitrate in drinking water include a case study of spontaneous abortions in women consuming high nitrate water (19–26 mg/L) from private wells (Morbidity and Mortality Weekly Report 1996).

While amassing experimental data suggest a role for nitrate in the formation of carcinogenic *N*-nitroso compounds, clear epidemiologic findings are lacking on the possible association of nitrate in drinking water with cancer risk. Ecologic studies have reported mixed results for cancers of the stomach, bladder, and esophagus (Barrett et al. 1998; Cantor 1997; Eicholzer and Gutzwiller 1990; Morales-Suarez-Varela et al. 1993, 1995) and non-Hodgkin lymphoma (Jensen 1982; Weisenburger 1993), positive findings for cancers of the nasopharynx (Cantor 1997), prostate (Cantor 1997), uterus (Jensen 1982; Thouez et al. 1981), and brain (Barrett et al. 1998), and negative findings for ovarian cancer (Jensen 1982; Thouez et al. 1981). Positive findings have generally been for long-term exposures at > 10 mg/L nitrate. Case–control studies have reported mixed results for stomach cancer (Cuello et al. 1976; Rademacher et al. 1992; Yang et al. 1998); positive results for non-Hodgkin lymphoma at > 4 mg/L nitrate (Ward et al. 1996) and colon cancer at > 5 mg/L (De Roos et al. 2003); and negative results for cancers of the brain (Mueller et al. 2001; Steindorf et al. 1994), bladder (Ward et al. 2003), and rectum (De Roos et al. 2003), all at < 10 mg/L. Cohort studies have reported no association between nitrate in drinking water and stomach cancer (Van Loon et al. 1998); positive associations with cancers of the bladder and ovary at long-term exposures > 2.5 mg/L (Weyer et al. 2001); and inverse associations with cancers of the rectum and uterus, again at > 2.5 mg/L (Weyer et al. 2001).

Exposure to low levels of antibiotics and other pharmaceuticals in drinking water (generally at micrograms per liter or nanograms per liter) represent unintentional doses of substances generally used for medical purposes to treat active disease or prevent disease. The concern is more related to possible cumulative effects of long-term low-dose exposures than on acute health effects (Daughton and Ternes 1999). A recent study conducted in Germany found that the margin between indirect daily exposure via drinking water and daily therapeutic dose was at least three orders of magnitude, concluding that exposure to pharmaceuticals via drinking water is not a major health concern (Webb et al. 2003). It should be noted that when prescribing medications, providers ensure patients are not taking incompatible drugs, but exposure via drinking water is beyond their control.

Endocrine-disrupting compounds are chemicals that exhibit biological hormonal activity, either by mimicking natural estrogens, by canceling or blocking hormonal actions, or by altering how natural hormones and their protein receptors are made (McLachlan and Korach 1995). Although very low levels of estrogenic compounds can stimulate cell activity, the potential for human health effects, such as breast and prostate cancers, and reproductive effects from exposure to endocrine disruptors, is in debate (Weyer and Riley 2001).

Workshop Recommendations

Priority research needs.

- Ecosystems monitoring: Systematic sustained studies of ecosystem health in proximity to large CAFOs are needed, including effects of input spikes during spills or flooding events.
- Toxicologic assessment of contaminants: Identification and prioritization of contaminants are needed to identify those that are most significant to environmental and public health. Toxicity studies need to be conducted to identify and quantify contaminants

(including metabolites), and to investigate interactions (synergistic, additive, and antagonistic effects).

- Fate and transport: Studies of parent compounds and metabolites in soil and water must be conducted, and the role of sediment as a carrier and reservoir of contaminants must be evaluated.
- Surveillance programs: Programs should be instituted to assess private well water quality in high-risk areas. Biomonitoring programs should be designed and implemented to assess actual dose from environmental exposures.

Translation of science to policy.

- Wastewater and drinking water treatment: Processes for water treatment must be monitored to ensure adequate removal or inactivation of emerging contaminants.
- Pollution prevention: Best management practices should be implemented to prevent or minimize release of contaminants into the environment.
- Education: Educational materials should be continued to be developed and distributed to agricultural producers.

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