Growing Tomatoes by Plasticulture

Over the past few decades, farmers have been experimenting with the use of a "plasticulture system" to increase their marketable vegetable yields. The plasticulture system is comprised of three main parts--the plastic mulch, raised beds, and a drip irrigation system. The plastic mulch is a 1.20 to 1.50 meter wide polyethylene plastic film, approximately 1.25 to 1.50 millimeters thick, and sold on 730 meter long rolls (North Carolina Cooperative Extension Service, 1996). The plastic mulch is available in either light colors or black. The color is chosen based on the temperature ranges typical to the growing season; light colors are used in the warm summer climate to keep the soil cooler and black is chosen in the cool spring climate to warm the soil. Raised beds are formed in long rows up to 183 meters long and 20 centimeters high (Virginia Cooperative Extension, 1997) before being covered with the plastic mulch. In plasticulture farming, 55% of the field is generally covered in the impermeable plastic. Drip irrigation lines are used in the plasticulture system to deliver both irrigation water and fertilizer directly to the plant root zone.

The plasticulture system has proven to have many advantages, as well as some disadvantages, over conventional farming techniques. The advantages and disadvantages are listed below (University of Georgia, 1999; Brooks and Sons, 1998; North Carolina Cooperative Extension Service, 1996).

Advantages:

1. Earlier spring production of the vegetables is possible because the black plastic has the ability to warm the soil by as much as 10ºF; this early production, in turn, can enable the farmer to get better prices at market for their vegetables.
2. The plastic mulch inhibits direct contact of the plant's leaves and fruit with the soil, producing a cleaner crop with less decay potential.
3. The plastic mulch directs rainfall from the fields, preventing it from percolating through the soil and washing the fertilizer nutrients away from the plant roots and to the groundwater table.
4. Weed growth is naturally controlled under the plastic because it does not transmit light.
5. The soil moisture content is more stable because the plastic mulch reduces evaporation of the soil water and also directs runoff from the raised beds, preventing saturation of the soil. Compared to a sprinkler system, the use of plastic mulch in combination with drip irrigation has been shown to cut water usage by 50-75%.
6. The plastic mulch and low pressure irrigation reduces compacting of the soil in the raised beds so it is loose and well aerated, which contributes to a healthier crops.

Disadvantages:

1. The initial start-up costs of a plasticulture system can be high because new farming equipment is often needed to lay the plastic, and the plastic mulch and drip tubing is often times replaced seasonally.
2. The system is more complicated to design, install, and manage than conventional farming techniques.

3. Drip irrigation, a main component of the plasticulture system, requires constant monitoring for leaks or clogs.
4. The removal and disposal of the plastic mulch at the end of the growing season is extremely time consuming.

Overall, farmers consider the advantages of using a plasticulture system to far outweigh the disadvantages, simply based on the increased vegetable production. Statewide, North Carolina has shown a triple increase in tomato production from the combined use of plasticulture and drip irrigation (North Carolina Cooperative Extension Service, 1996).

Over the past decade, farmers on the Eastern Shore of Virginia have begun using the plasticulture system to aid in increasing their marketable vegetable production, specifically tomatoes. Two overlapping growing seasons are used in Virginia tomato farming, the first beginning in April, the second beginning in late June. When using the plasticulture system, the fields must be prepared prior to transplanting the tomato plants. The field preparation begins in March for the first growing season and in May for the second growing season (Klawiter, 1998). The preparation process consists of the following:

1. The fields are deeply tilled to allow the tomato plants maximum root growth.
2. Raised beds are formed in long rows up to 183 meters long and 20 centimeters high (Virginia Cooperative Extension, 1997).
3. The drip irrigation/fertilizer lines and plastic mulch are laid. Black plastic mulch is used to warm the soil for the first growing season, and light colored plastic mulch is used to cool the soil for the second season.

4. Farm machinery is used to place evenly spaced holes in the plastic.
5. Farm machinery is also used to drive 1.2 meters high wooden stakes 0.3 meters deep into the ground (Geisenberg and Stewart, 1986).

After the preparation process, the tomato plants, begun as seedlings in a greenhouse, are transplanted into the holes in the plastic mulch near the wooden stakes (Geisenberg and Stewart, 1986). For the first growing season, this occurs in April and for the second season in late June. During their growing season, the plants are tied to the stakes on four separate occasions (Klawiter, 1998 citing Gayle, 1996) and regularly pruned to maximize fruit production (Virginia Cooperative Extension, 1997). Tomatoes are manually harvested three times during the growing season (Klawiter, 1998 citing Gayle, 1996). The final harvesting occurs in July for the first season and late September for the second. Following the final harvest, the remaining plant material is dessicated and removed as a disease control measure; the plastic mulch, all stakes, and string are also removed (Geisenberg and Stewart, 1986). Thicker plastic mulches and drip irrigation/fertilization tubing can be re-used the following year if they are placed indoors in the winter; however, many farmers replace these every year (Virginia Cooperative Extension, 1996).

Tomato Plant Diseases

Tomato plants are subject to a variety of diseases that may impact both crop production and marketability. Two of the most devastating diseases to tomato plants are bacterial speck, *Pseudomonas syringae* pv. *tomato* and bacterial spot, *Xanthomonas campestris* pv. *vesticatoria*. Farmers can identify both of these diseases through the dessication and death of the plant leaves (Goode, 1980).

Bacterial speck is characterized by swelling black or brown spots, roughly a few millimeters in diameter; these spots can be found on both the leaves of the plant as well as the fruit. Typically the tissue surrounding the spots is yellowish in color. The cool and moist climate of the spring favors an infection of bacterial speck; the on-set of this disease can take less than 24 hours in overly moist conditions (Blancard, 1994; Jones *et al.*, 1991; Watterson, 1986). An interesting aspect of bacterial speck is that infection can seem to have disappeared before re-emerging with an overall crop loss of more than 25% of the yield (University of California, 1985).

Bacterial spot is very similar to bacterial speck in that it is also characterized by black spots, surrounded by light tissue, on the leaves and fruit of the plant. However, the spots vary significantly in that they are dry and scabby in nature. The spots generally begin at one-millimeter in diameter and can continue to grow, reaching one-centimeter in diameter before falling off of the plant. Like bacterial speck, bacterial spot is favored in moist conditions, but differs in that it thrives in warmer climates (Blancard, 1994; Jones *et al.*, 1991; Watterson, 1986).

Several measures have been used to control outbreaks of these devastating diseases in tomato plants. The use of plastic mulch has proven beneficial in controlling both bacterial speck and bacterial spot by maintaining more stable soil moisture conditions (Klawiter, 1998 citing Gayle, 1996; Geisenberg, 1986). Also, farmers attempt to control the negative impact of environmental factors by scheduling planting and harvesting when conditions for disease transmission is less favorable. In addition, some varieties of tomato plants have been developed with resistance to bacterial speck; although, no marketable variety with resistance to bacterial spot presently exists.

Farmers also rely on a plant's natural ability to resist the diseases; resistance is determined by variety, the presence of light and moisture, temperature, surrounding gases, and micro-injuries to the plant through conventional tomato farming practices (Watterson, 1986). As a final measure to ward off disease, farmers also apply crop protectants.

Copper as a Crop Protectant

In tomato farming, copper-based crop protectants serve as an effective bactericide and fungicide against a number of diseases, including bacterial spot and bacterial speck (Blancard, 1994; Jones *et al.*, 1991; Watterson, 1986). The copper is available as a number of different chemical compounds including copper hydroxides, basic sulfates, oxides, and oxychlorides. Copper is also available in several formulations including microflows, wettable powders, and water dispersible granules (Meister, 1996).

Depending on the chemical form of the copper, the properties of the crop protectant may differ dramatically. Copper oxides, oxychlorates, and sulfates are very insoluble in water as well as most organic solvents. The solubility of copper hydroxides range from insoluble to highly soluble depending on the formulation (Griffin, 1996 d, e, f). Wettable powders are insoluble and require constant mixing to remain suspended in solution, whereas microflows contain the copper already in solution (Virginia Cooperative Extension, 1996).

During the three-month growing period, between 1.70 and 3.40 kilograms of free copper per hectare per week is recommended to ward off disease outbreaks; the exact quantity required depends on the formulation, size of the crop, rainfall, and severity of the disease (Griffin, 1996 a, b, c). As an alternative, the Virginia Cooperative Extension

(1997) recommends a combination of one pound of free copper and one-and-a-half pounds of the fungicide mancozeb per acre per week. Typically, crop protectants are applied through an overhead sprinkler system, traveling sprayer, or aerial spraying (Griffin, 1996 a, b, c). However, applying the crop protectant through a ground-based traveling sprayer has been proven the best method for coating the entire plant (Klawiter, 1998 citing Gayle, 1996).

Copper's toxic nature to aquatic life has led manufacturers of copper-based crop protectants to advise farmers against applying directly to waterbodies or allowing entrance via runoff (Griffin, 1996 a, b, c). The North Carolina Agricultural Extension Service advises farmers not to apply crop protectants to wet ground because of the runoff potential. It is also recommended that sprayers not be used to apply crop protectants when the winds exceed five miles per hour due to the potential for drift into the nearby waterways runoff (Griffin, 1996 a, b, c; North Carolina Agricultural Extension Service, 1988).

<u>Copper Toxicity</u>

Copper is considered an essential element for sustaining natural biological processes in aquatic organisms; however, in excess amounts this metal can be lethal. Copper, silver, and mercury are considered some of the most toxic metals to aquatic life, with LC_{50} values ranging from 5 to 100,000 µg/L. Copper is known to be toxic at low doses (Brezonik *et al.*, 1991; Hodson *et al.*, 1979). Precise metals toxicity levels are difficult to determine, as they depend on a variety of factors including the species, size of the organism, age and stage of development (Eisler, 1979). The form of the metal also affects toxicity (Sanders *et al.*, 1991). Metals do not need to have an immediate effect on aquatic life to be considered toxic as some species are capable of bioaccumulation, causing a delayed reaction (Luoma and Carter, 1991; Eisler, 1979).

The ability of an organism to tolerate an environment with copper concentrations in excess of their natural requirements varies dramatically from species to species and even amongst individuals of the same species. In general, larger organisms have the capability to handle higher copper concentrations. Age and developmental stage also impact toxicity, with older organisms having a tendency to be more tolerant (Eisler, 1979). A few exceptions exist including eggs which can be more resistant than the young (Bryan, 1971) and reproducing adults with lowered resistance as their energy stores are depleted (Luoma and Carter, 1991). Some organisms also have the ability to adapt to an environment with higher copper concentrations (Eisler, 1979).

Copper may exist in many physical as well as chemical forms in waterbodies, affecting its toxicity to aquatic life. Physically copper may be found sorbed to bottom or suspended sediment, bound to suspended biota such as algae, or dissolved in the water column or pore water (Burgess *et al.*, 1993). In general it has been found that the distribution of pesticides between the solid and liquid phase often fits the Freundlich isotherm, $x/m = KC^{1/n}$ where x/m is the mass adsorbed per mass of sorbent, C is the equilibrium adsorption concentration, and K and n are constants (Saltzman and Yaron, 1986).

Pesticides may sorb to soils via many different mechanisms including Van der Waals forces, hydrogen bonding, hydrophobic bonding, ion exchange, charge transfer, ligand exchange, and chemisorption. Van der Waals forces act to force short range interactions between ions, maintaining dipole-dipole or induced dipole-dipole interactions. Hydrogen bonding, like Van der Waals forces, is also a dipole-dipole interaction; it differs, however, in that hydrogen atoms are used to bridge two electronegatively charged atoms. Hydrogen bonding is a stronger bond than Van der Waals forces, but weaker than ionic bonds. Hydrophobic bonds exist between nonpolar organic compounds in which adsorption occurs on the surface of organic matter in the forms of fats, waxes, and resins. Ion exchange, the typical sorption mechanism for cationic pesticides, is a simple cation exchange process between the pesticide and the soil particle. Charge transfer occurs from electrostatic interactions in which electrons are transferred from the pesticide, serving as an electron donor, to organic matter that serves as the electron acceptor. Ligand exchange occurs when the pesticide is a stronger chelating agent than the ligand on the soil particle and simply replaces it. Chemisorption refers to a covalent chemical bond that forms between the pesticide and the soil particle in an exothermic process (Cresser, Killham and Edwards, 1993).

Most sorption of pesticides to soil occurs through adsorption to organic matter. However, adsorption is also correlated to the cation exchange capacity of the soil, pH of the soil, moisture content of the soil, and the characteristics of the pesticide.   Maximum sorption of a pesticide to soil occurs when the soil pH value is close to the pK value of the pesticide for adsorption (Saltzman and Yaron, 1986 citing Weber *et al.*, 1969). Soil moisture plays an important role in the sorption of the pesticide to the soil since the pesticide is typically carried in solution; therefore, the water in the soil directly determines the accessibility of the pesticide to the soil's adsorption sites. The characteristics of the pesticide such as molecular structure, shape, size, acidity or basicity, water solubility, and charge distribution also control the sorption (Saltzman and Yaron, 1986).

Determining which physical form of copper is toxic to an organism can be difficult (McIntosh, 1991). Burgess *et al.* (1993) reported that toxicity associated with pore water was not associated with anthropogenic heavy metals, but more likely low molecular weight hydrophilic organics, ammonia, and/or hydrogen sulfide. It has also been suggested toxicity increases as the particle size to which the copper is sorbed decreases, implying copper associated with the suspended sediment is more toxic than that sorbed to bottom sediment (Burgess *et al.*, 1993).

Toxicity of copper in the water column greatly depends on the chemical form. Copper has the ability to form a wide variety of complexes and bonds. Copper can form both inorganic and organic complexes. Ionic bonds may be formed with hydroxides, sulfides, nitrates, and chlorides (Meador, 1991).

In general, the formation of complexes decrease the toxicity of copper to aquatic organisms. Inorganic complexes, such as those formed with calcium and magnesium, reduce copper toxicity by competing for the same binding sites in organisms; therefore, harder waters decrease the potential for toxicity of copper to aquatic life (Brezonik *et al.*, 1991). Increased alkalinity, associated with harder waters, also contributes to reducing copper toxicity as the carbonate ion forms complexes (Snoeyink and Jenkins, 1980). Organic complexes are also considered to decrease toxicity; however, bioaccumulation typically occurs, causing a potential threat to the organism in the future (Brezonik *et al.*, 1991). Possible compounds of copper-organic complexes include those formed with phthalic, pyruvic, glycolic, acetic, formic, humic, and fulvic acids (Leckie and Davis, 1979).

The presence of copper as an ion also affects the toxicity, increasing it beyond that of total copper (Zamuda and Sunda, 1982); ionic copper concentrations are controlled by the pH of the natural water. Copper concentrations in low pH water are considered more toxic to aquatic life as the cupric ion concentration is driven upward (Guthrie and Perry, 1980). However, as the cupric ion concentration increases in natural waters, so does the hydrogen ion; hydrogen ions compete with the copper ion for binding sites on organisms, possibly contributing to a decrease in toxicity (Meador, 1991). Waters with the same ionic copper concentration and a pH above seven are considered more toxic than low pH waters, indicating the presence of other toxic copper complexes (Meador, 1991). The formation of soluble salts from the copper ion, such as copper sulfate, nitrate, and chloride, is also a possibility.

Copper can form ionic bonds with hydroxides and sulfides, forming a less soluble compound where precipitation is even possible (USEPA, 1985). The reported solubility of cupric hydroxide ranges quite significantly over three orders of magnitude, making it very difficult to predict its bioavailability. Review of the literature suggests this wide range in solubility has been attributed to surface area effects (Baes and Mesmer, 1976; Schindler *et al.*, 1965) and/or aging (Patterson *et al.*, 1991). Hidmi and Edwards (1999) reported the log K solubility for fresh cupric hydroxide as 10.2 ± 0.2 with ΔH of +10.42. Precipitation was reported to occur at pH values of 7.0, 8.0, and 9.0 in a 1 mM $Cu(NO_3)_2$ solution. They also concluded that the differences in reported solubility are a result of solid age, difficulties in attaining equilibrium, and the initial formation of cupric nitrate solids in nitrate electrolyte containing solutions (1999).

Despite all of the possible copper forms, both physical and chemical, that affect its toxicity, bioavailability is ultimately the determining factor in copper toxicity to an organism. Bioavailability is defined as the copper that is "available for uptake by a living organism" (Greer, 1995); however, bioavailable copper is not a defined quantity, but rather depends on each individual or species. Toxicity appears to occur as free ions bind to key receptors, impairing natural biological processes (Renner, 1997).

The toxicity of copper to a variety of shellfish important to the Eastern Seaboard economy has been established, including data for the Bay scallop, American oyster, and hard clam. The Bay scallop, *Argopecten irradians*, shows retardation of soft tissue growth and even detachment at an ionic copper concentration of 5.8 µg/L and the 42-day 50% lethal concentration (LC_{50}) is 9.3 µg/L (Pesch *et al.*, 1979). At a concentration of 32.8 µg/L added copper, the American oyster, *Crassostrea virginica*, exhibits slower development, growing to only 67.7% of the size of a control group. The lethal concentrations vary between embryos and larvae. The LC_{50} for embryos is 103 µg/L added copper, determined by the number that developed into straight-hinge larvae within normal time limits. The 12-day LC_{50} for larvae is considerably lower, at only 32.8 µg/L (Calabrese *et al.*, 1973). Experiments with the hard clam, *Mercenaria mercenaria*, show an LC_{50} between eight and ten days at a concentration of just 16.4 µg/L and significantly less growth at only 51.7% of that of the control group (Calabrese *et al.*, 1977). LaBreche (1998) reported physical deformities, loss of activity, and even death in larval hard clams exposed to copper in concentrations of 4-8 µg/L for a one to three day period.

Regulations on Copper in Waterways

Overwhelming public concern for the country's water pollution problems led to the Clean Water Act of 1972. This act was designed to restore and maintain the quality of the nation's waters. Although it has been amended through the years, the basic framework establishing standards for pollution discharge, the use of best achieveable pollution control strategies, and provisions for financial assistance has remained intact.

The most dramatic amendments came five years later in the legislation known as the Clean Water Act Amendments of 1977, which focused on strengthening the controls of toxic pollutants. Section 307(a) of these amendments created a list of 65 toxic priority pollutants. Section 304(a) required the Environmental Protection Agency (EPA) to publish and periodically update criteria regarding the health and welfare effects on the priority pollutants. Section 303 of the Clean Water Act Amendments granted state governments authority to implement water quality standards as long as they were in keeping with those federally established.

In 1987, the Water Quality Act was passed and renewed interest in achieving the goals established by the Clean Water Act and its amendments. The Water Quality Act focused on supporting state and local pollution control efforts, created revolving loan funds, addressed urban runoff, and created nationally important estuary protection programs.

Copper is considered a priority pollutant and federal water quality standards have been established for both saltwater and freshwater (USEPA, 1985). In 1985, the criteria for copper in saltwater was set at 4.0 µg/L total recoverable copper as a 24-hour average not to ever exceed 23 µg/L total recoverable copper. For freshwater it was set at 5.6 µg/L total recoverable copper as a 24-hour average not to exceed a concentration established by the following formula:

$$\text{Maximum 24-hour average, µg/L total recoverable copper} = \exp[0.94(\ln(\text{hardness})) - 1.23]$$

Hardness is expressed as mg/L $CaCO_3$ with a minimum concentration of 25 mg/L and a maximum concentration of 400 mg/L. Under Section 303 of the Clean Water Act Amendments of 1977, the State of Virginia chose to adopt a water quality standard more stringent than the federally established standard.

In Virginia, the saltwater criteria were set at 2.9 µg/L dissolved copper for both the acute and chronic case. The acute case was established as a one-hour concentration not to be exceeded more than once every three years and the chronic as a four-day average not to be exceeded more than once every three years. The freshwater criteria for copper in Virginia established different acute and chronic dissolved copper values based on equations (Virginia Department of Environmental Quality, 1992). For the acute case, the formula for the one-hour average dissolved copper concentration in freshwater, not to be exceeded more than once every three years, is as follows:

$$\text{One-hour average, µg/L dissolved copper} = \exp[0.9422(\ln(\text{hardness})) - 1.464]$$

For the chronic case, the formula for the four-day average in freshwater, not to be exceeded more than once every three years, is as follows:

$$\text{Four-day average, µg/L dissolved copper} = \exp[0.8545(\ln(\text{hardness})) - 1.465]$$

In both of these formulas, hardness is expressed as mg/L $CaCO_3$ with a minimum concentration of 25 mg/L and a maximum concentration of 400 mg/L.

In 1995, the United States Environmental Protection Agency published a draft, updating the ambient water quality criteria for copper in saltwater to a more stringent level. The acute concentration was changed to 4.8 µg/L dissolved copper as a 24-hour average not to be exceeded more than once every three years. The chronic concentration was modified to 3.1 µg/L dissolved copper as a four-day average not to be exceeded once every three years. These changes were still less stringent than those established by the Virginia Department of Environmental Quality.

Programs to Minimize Non-Point Source Pollution

Though exact percentages differ among sources, an estimated 50-80% of the nation's pollution problems in water result from non-point source releases. With this in mind, many formal programs, in addition to the state and federally established water quality standards, have been established to abate the non-point source pollution problems. The Food Security Act of 1985 was established in an effort to reduce solids loadings to waterways by providing funds to take erodible lands next to waterways out of production (Novotny and Olem, 1994). The Chesapeake Bay Program, founded under the Water Quality Act of 1987, was established to help restore the bay to its natural state. The Water Quality Act also provided for the National Estuary Program, under which NOAA's Sea Grant supports research in estuarine areas. In 1990, the Coastal Zone Act Reauthorization Amendments were established to reduce polluted runoff water in the coastal areas of 29 states, including Virginia.

Less formal, voluntary programs have also been established to reduce non-point source pollution. These voluntary programs, which call for the incorporation of best management practices (BMPs), often compensate farmers with funds provided by the government and provide free technical advice. Today, many different best management practices exist, varying in their cost to implement and maintain, effectiveness, and pollution control mechanism. It is important to note that farmers are not restricted to implementing a single best management practice. Systems of best management practices are the most effective.

Soil erosion is considered the largest contributor of non-point source pollution to waterways, severely impacting their storage capacity. Several best management practices have been developed to control soil losses including grass and vegetative filter strips, sedimentation control, environmentally-conscience tillage practices, and crop rotations. Funds have even been provided to take critical (highly erodible) lands out of production (Novotny and Olem, 1994).

Vegetative and grass filter strips are used to protect water quality by serving as a buffer between the agricultural field and the body of water. Filter strips consist of closely spaced vegetation such as sod, bunch grasses, or small grain crops. They operate by slowing down the velocity of the runoff and allowing the sediment and any sorbed pollutants to settle out. While the effectiveness of filter strips depends on their width and vegetative cover, they are reported to remove between 35-90% sediment and sediment-bound nitrogen; however, they are much less effective in reducing phosphorus, fine sediment, and soluble nutrients (Novotny and Olem, 1994).

Sedimentation basins are used to reduce the loading of sediment to local waterbodies by either detaining or retaining the runoff. In detention ponds, the runoff is collected and permanently stored; vegetation often grows in these ponds. Retention ponds differ in that they are designed to only hold the runoff for a period of time. Both ponds operate on the same key principle of stopping the flow of runoff to allow for the settling and capture of the heavier particles. The effectiveness of sedimentation basins varies from 40-87% removal of the incoming sediment. Sedimentation control is also considered an effective means of reducing nitrogen and phosphorus sorbed to the soil, with 30% and 40% reductions, respectively. However, sedimentation control is not considered effective in the removal of compounds in the dissolved state (Novotny and Olem, 1994).

Several different tillage and land use practices also exist for reducing erosion of agricultural land. A variety of tillage practices, including conservation tillage and no-till planting, were developed on the premise of maintaining a vegetative cover on the fields to reduce soil loss. Conservation tillage is the practice of leaving at least 30% of the field, after planting, covered with crop residue. The soil is also only tilled to the degree needed for preparing a seedbed. Conservation tillage has been reported to reduce soil loss by 30-90%, total phosphorus by 35-85%, and total nitrogen by 50-80%. No-till planting is the process of planting seeds into untilled soil while retaining prior plant residue. This practice is considered very effective in reducing erosion from agricultural fields. Crop rotation, the practice of periodically changing the crops grown in an area, is also considered an effective means for reducing non-point source pollution. Crop rotation is most effective when at least two years of grass or legumes are planted in a four-year rotation. The effectiveness of crop rotation is estimated by averaging the reduction for each year in the rotation. Estimates of nitrogen and phosphorus reduction have been reported at 50% and 30% annually, respectively (Novotny and Olem, 1994).

Reducing the loss of crop protectants from agricultural fields is also a key area in non-point source pollution control. Several strategies for their reduction have been developed; these include improving the efficiency of application, integrated pest management strategies, soil and water conservation practices, and using alternative crop-protectants (North Carolina Cooperative Extension Service, 1984). Using metals-based crop protectants require different best management methods including the selection of plants requiring less metals usage to ward off disease, reuse of runoff as irrigation water, physical or biological filtration of runoff, integrated pest management, and creation of artificial wetlands for treating runoff (Novotny and Olem, 1994).

Integrated pest management, known as IPM, utilizes a combination of practices to control insects, weeds, and diseases while simultaneously reducing pollution. Selecting resistant crops, using crop rotations, modifying crop planting dates, and managing pesticide applications, are some components of an IPM system. This system operates on the principle of decreasing the amount of pesticide or chemical crop-protectant required and by selecting the least toxic, mobile, or persistant chemical to decrease pollution potential. The effectiveness of an IPM system is site specific depending on the type of soil as well as pesticide and crops used (Novotny and Olem, 1994).

Wetlands function by naturally removing, transferring, or storing pollutants as the runoff slowly passes through. They operate on the same principle as most other runoff treatment processes, with the idea of slowing down the transport of runoff and allowing sediment to settle out. They are between 80-90% effective in reducing sediment loading. Wetlands vegetation are also effective in reducing nitrogen and phosphorus levels to between 40-80% and 10-70%, respectively. Wetlands are also considered to have potential for reducing other pollutants such as chemicals and metals as they are either sorbed to the soil, taken up by plants, or degraded by biological activity (Novotny and Olem, 1994).

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The use of crop protectants by farmers in coastal regions has impacted water quality. Scott *et al.* (1990) reported elevated levels of endosulfan, an insecticide, in sediments taken from an unnamed tidal tributary near agricultural land on Kiawah Island, located just south of Charleston, South Carolina. Elevated loadings of organonitrogen and organophosphorus pesticides in the Chesapeake Bay were directly correlated to the time and rate of field applications (Foster and Lippa, 1996). The use of copper-based crop protectants on tomato fields, planted in plasticulture, has been linked to deteriorated water quality on the Eastern Shore of Virginia (Brady *et al.*, 1999; Dietrich *et al.*, 1999; Klawiter, 1998; Dietrich *et al.*, 1996).

In these coastal areas, extremely short distances often exist between the agricultural fields and water bodies, increasing the probability that a pesticide will reach the waterway via runoff from the fields. One agriculture practice that has greatly enhanced the potential of a pesticide to enter a waterway via runoff is plasticulture.

The plasticulture system is comprised of three main parts: the plastic mulch, raised beds, and a drip irrigation system. The plastic mulch is 1.20 to 1.50 meter wide polyethylene plastic film, approximately 1.25 to 1.50 millimeters thick, and sold on 730 meter long rolls. Twenty centimeter high raised beds are formed in rows up to 183 meters long (Virginia Cooperative Extension, 1997) before being covered with the plastic mulch. Drip irrigation lines, placed under the plastic mulch, are used in the plasticulture system to deliver both irrigation water and fertilizer in low pressure, concentrated doses.

The main advantage of the plasticulture system is that it can significantly increase marketable vegetable yields. Other advantages include controlled soil moisture, less nutrients and fertilizer loss via runoff, naturally controlled weed growth, and a healthier crop (University of Georgia, 1999; Brooks and Sons, 1998; North Carolina Cooperative Extension Service, 1996).

A disadvantage to plasticulture farming is the increased runoff volumes that are a result of the significantly reduced permeability of the agricultural fields. In plasticulture, 55% of the field is generally covered in the impermeable plastic. In combination with the soil between the plasticulture rows being compacted by farm equipment, plasticulture fields have been shown to have reduced permeability to rain. Other disadvantages to plasticulture farming include increased cost to the farmer and more time to install and maintain than traditional farming practices (University of Georgia, 1999; Brooks and Sons, 1998; North Carolina Cooperative Extension Service, 1996; Scott *et al.*, 1990).

Plasticulture has been quite effective in improving tomato crop yields and is considered a best management practice based on its ability to control herbicide and fertilizer use as well as disease outbreaks related to watering, such as bacterial spot and bacterial speck (Meister, 1996; Blancard, 1994; Jones *et al.*, 1991; Watterson, 1986). Plasticulture works to fight disease outbreaks by forcing rainwater off of the tomato fields and leaving only controlled irrigation as the water source for the plants; however, plasticulture farming cannot completely control these diseases and bactericides and fungicides are still needed.

Copper-based crop protectants are commonly used for tomato plants because of their ability to treat many diseases including bacterial spot and bacterial speck (Meister, 1996; Blancard, 1994; Jones *et al.*, 1991; Watterson, 1986). Between 1.70 and 3.40 kilograms of copper-based crop protectants per hectare per week are applied throughout the three month growing season, generally through a traveling sprayer (Griffin, 1996 a, b, c). Farmers determine the amount of crop-protectant to apply based on the copper content in the protectant, size of the crop, severity of the disease, and rainfall amounts. Because rainwater washes the copper-based crop protectant off the plant, common practice in tomato farming is to reapply the protectant after rainfall events.

Copper may exist in many physical as well as chemical forms in water bodies. Physically, copper may be found sorbed to bottom or suspended sediment, bound to suspended biota such as algae, or dissolved in the water column or pore water (Burgess *et al.*, 1993). Chemically, copper has the ability to form a wide variety of both organic and inorganic complexes as well as ionic bonds with hydroxides, sulfides, nitrates, and chlorides (Meador, 1991).

The chemical speciation of copper will affect its toxicity. Free copper (II) ion is considered highly toxic, while the formation of complexes decreases the toxicity of copper to aquatic organisms (Zamuda and Sunda, 1982). Copper is considered an essential element for sustaining natural biological processes in aquatic organisms; however, excess amounts of this metal can be lethal. Copper, silver, and mercury are considered some of the most toxic metals to aquatic life, with LC_{50} values ranging from as low as five up to 100,000 μg/L; copper is known to be toxic at low doses (Brezonik *et al.*, 1991; Hodson *et al.*, 1979).

Most sorption of pesticides to soil occurs through adsorption to organic matter. Adsorption is also correlated to the cation exchange capacity of the soil, pH of the soil, moisture content of the soil, and the characteristics of the pesticide. The characteristics of the pesticide which affect sorption are molecular structure, shape, size, acidity or basicity, water solubility, and charge distribution (Saltzman and Yaron, 1986).

For conventional agriculture, it is estimated that less than one to two percent of the applied pesticide leaves the agricultural field via runoff (Volger, 1997; Wauchope, 1996). Volger (1997) reported data for endosulfan, while Wauchope (1996) reported data for pesticides in general. Likewise, less than one-tenth to one percent of the applied pesticide is estimated to leach through the soil (Flury, 1996). In a study performed at a conventional agricultural site on Virginia's Eastern Shore, land applied nitrogen, alachlor, metalochlor, atrazine, and cyanazine were demonstrated to be transported through sandy/loamy soils into the groundwater; eventually, these nutrients and pesticides entered the Chesapeake Bay via submarine groundwater discharge (Gallagher *et al.*, 1996). Because pesticides can travel in runoff and groundwater from an agricultural field to an adjacent waterway, there exists a need for watershed protection and concern for the toxic effects of the pesticides to aquatic life.

Many programs exist to reduce nonpoint source pollution. One voluntary program, in which farmers can participate, is the incorporation of best management practices (BMPs). In agriculture, two key pollutants in need of control are soil erosion and crop protectant washoff.

Best management practices that have been developed to reduce soil losses include sedimentation control, grass and vegetative filter strips, changes in tillage practices, and crop rotations. Their effectiveness ranges from a 30-90% reduction in soil loss. However, sedimentation basins and filter strips have the added advantage in reducing pollutants sorbed to the soil. Sedimentation control has shown a 30% decrease in nitrogen and 40% decrease in phosphorus sorbed to the soil (Novotny and Olem, 1994). *Scott et al.* (1991) reported a greater than 90% reduction in endosulfan concentrations in estuarine surface water from the incorporation of a retention pond on an agricultural field. While filter strips are effective in reducing sorbed nitrogen, 35 to 90% reductions have been reported, they are much less effective in reducing sorbed phosphorus (Novotny and Olem, 1994).

Another BMP commonly used in agriculture is integrated pest management (IPM) which uses a combination of practices to reduce the amount of crop-protectant required and select the least toxic, mobile, or persistant chemical to decrease pollution potential. The effectiveness of an IPM system is site specific, depending on the type of soil as well as the pesticide and crops used (Novotny and Olem, 1994).

One area in which a need for BMPs exist is the Eastern Shore of Virginia, where plasticulture farming and crop protectant usage have been directly linked to deteriorations of water quality standards. Research in this area has shown copper concentrations in agricultural runoff samples up to 1450 μg/L (Dietrich *et al.*, 1996), with typical concentrations of 20-250 μg/L dissolved copper (Brady *et al.*, 1999). Concentrations up to 126 μg/L dissolved copper in tidal creeks impacted by tomato plasticulture have also been reported (Klawiter, 1998), which are well in excess of the 2.9 μg/L dissolved copper water quality standard for saltwater established by the State of Virginia (Department of Environmental Quality, 1992).

The influx of copper-based crop protectants into tidal creeks and estuaries is a potential threat to the aquaculture industry, which pump estuarine water directly into their hatcheries. Commercially important shellfish are very sensitive to copper, experiencing deformities and death at low concentrations. Calabrese *et al.* (1977) reported an eight to ten day LC_{50} value of 16.4 μg/L copper for the larval hard clam and a twelve day LC_{50} value of 32.8 μg/L copper for the larval American oyster, *Crassostrea virginica*. LaBreche (1998) reported physical deformities, low activity, and even death to larval *Mercenaria mercenaria* at concentrations of 4-8 μg/L copper from one to three days of exposure.

The research reported here is part of a larger project in which the agricultural runoff from simulated tomato plasticulture fields was input to simulated estuaries containing plants and aquatic life. This research project focused on evaluating the fate of copper-based crop protectants, applied to plasticulture fields, following rainfall events in a controlled greenhouse environment. The specific objectives of this research project were as follows:

- Simulate tomato plasticulture fields using soil bins with Bojac Sandy Loam soil on a greenhouse-scale. Bojac Sandy Loam is a common soil type found in Virginia coastal regions.
- Compare the volumes of runoff and groundwater leaving the simulated tomato fields.
- Determine the fate and distribution of applied copper-based crop protectants.
- Evaluate the partitioning (sorbed versus dissolved) of copper in the runoff and groundwater.
- Evaluate the effectiveness of sedimentation control as a best management practice for reducing copper loadings to estuaries and tidal creeks.

All chemicals, sample containers, and glassware were purchased from Fisher Scientific (Raleigh, North Carolina). All containers used in the collection of samples and all glassware, funnels, and pipettes used in the analysis were soaked in 10% trace metal grade nitric acid for 8 hours, rinsed with distilled water three times, rinsed with Nanopure® water three times, and allowed to air dry.

Materials used in the construction of the simulated tomato fields were purchased from local suppliers. The Bojac Sandy Loam topsoil was obtained from the Gerald M. Moore & Son, Inc. asphalt plant, located in the town of Exmore on the Eastern Shore of Virginia.

Setup of Greenhouse-Scale Simulation

Five cubic yards of Bojac Sandy Loam topsoil were transported to a greenhouse on the Virginia Tech campus. At depths ranging from zero to 27 inches, Bojac Sandy Loam is characterized by a strong brown color, a sandy loam or loam texture with distinct clay bridges between the sand grains, and moderately rapid permeability with water draining at two to six inches per hour. The Bojac Sandy Loam used in this research was determined to have a baseline total copper content of 4.25 ± 0.18 mg/kg soil, a pH of 6.5 ± 0.06, an organic matter content of 1.2 ± 0.06%, and plant available copper concentration of 0.8 ± 0 mg/L.

Three separate wooden bins were constructed for the greenhouse-scale simulation of the coastal tomato fields. Each bin was constructed with 1.2 centimeters thick plywood supported in a 1.2 by 0.9 meters wooden structural frame to have dimensions of 2.1 by 1.2 by 0.9 meters. The interior of the bins was coated with KOOLPATCH® White

Patching Cement and KOOLSEAL® Elasometric Roof Coating to prevent leaks and warping. One bin was designed to contain a conventional plasticulture field, the second a field in plasticulture with sedimentation control, and the third a control field without plasticulture or copper additions. A schematic of the simulated tomato fields is presented in Figure 1.

Before filling the bins, a plastic garden-hose connection and nozzle was attached to the bottom corner of each bin to serve as the outlet for the groundwater collection system. The filling process was comprised of constructing an underdrain system, followed by adding soil to each bin, applying plastic mulch to two of the bins, and developing the runoff capture system. The underdrain was constructed by laying garden liner on the bottom of the boxes followed by adding ten centimeters of pre-washed Johnson's River Pebbles. Another layer of garden liner was placed on top of the pebbles. Once the liner was in place, approximately one-and-a-half tons of soil was added to each bin, spread out evenly, and shaped into two 20 centimeter high mounds with a compacted center. Drip irrigation lines were run along the length of the mounds. The soil was then allowed to settle for about a week before plastic mulch was applied to the mounds of two simulated fields; the third field was left with a natural soil cover to serve as the control field.

Patio hybrid tomatoes, purchased from Wetsel Seed Company of Virginia, were planted 35 centimeters apart in two rows for a total of fourteen plants per bin. For the simulated plasticulture fields, the tomatoes were planted directly through the plastic mulch. Over the duration of the growing season, the tomatoes were tied to wooden stakes with twine. The tomato plants were watered by a drip line irrigation system that ran along the length of each mound under the plastic mulch. The irrigation ran off a main line connected to a tap water supply at the greenhouse and was designed to water each of the fields at a uniform rate. The tomato plants were treated with Miracle Grow® and Epsom Salts during the growing season.

Microsperse® brand crop protectant, manufactured by Microflow, Inc. of Florida, was regularly applied to the tomato plants at a target concentration 1190 mg/L free copper. Microsperse® consisted of 53% metallic copper as copper oxychloride. The crop protectant was applied to the plants with a commercial hand-held sprayer. Table 1 outlines the application dates of the crop protectant in relationship to the dates of the simulated rain events. The control field, the non-plasticulture field, was not sprayed with copper.

A rainfall simulator was constructed with a network of 1.2-centimeter (0.5-inch) PVC pipes, a Master Plumber® portable submersible utility pump, and a Cole Parmer® flow meter with a 189 liters per hour capacity. Numerous 0.08-centimeter (1/32-inch) holes were drilled every 3.8 centimeters along the length of each PVC pipe to create 6.11 $\pm$ 0.25 mm diameter raindrops. The final dimensions of the rainfall simulator were 2.1 meters long by 1.2 meters wide. A rainfall intensity of 6.4 centimeters per hour (2.5 inches per hour), with a duration of twenty minutes, was used for each field and each rain simulation. A total of 163 liters of distilled water were applied to each field during each rainfall.

<u>Sample Collection Procedure</u>

Runoff and groundwater samples were separately collected at four, ten, and twenty minutes into the rainfall event. The remaining runoff and groundwater were collected in separate 19-liter plastic buckets from which three composite samples were taken. All runoff and groundwater samples were analyzed for total suspended solids as well as total, sorbed, and dissolved copper.

After the collection of samples, the remaining composite runoff from the simulated plasticulture field with sedimentation control was settled for five days. Runoff and groundwater from each field were then added to marine mesocosms for toxicity evaluation as reported by Cheadle *et al.* (1999).

At the end of the project, soil cores were taken from each tomato field to determine the distribution of sorbed copper on the fields. A total of fifteen soil cores were taken from each field, seven down the center of the field and four along each of the two mounds. Soil cores were collected and analyzed for the soil depth intervals in centimeters of 0-2.5, 2.5-5, 5-10, 10-15, and every five centimeters for the remaining field depth.

Samples Analysis Procedures

The total suspended solids analysis was performed according to Standard Methods for the Examination of Water and Wastewater, number 2540D (1995). Fisherbrand® Glass Fiber Circles were used for analysis. Total copper was analyzed according to EPA Method 3020A (USEPA, 1997). The copper sorbed to the soil was determined by digesting the glass fiber circles used to measure the suspended solids concentration according to EPA Method 3050A (USEPA, 1997). The dissolved copper analysis was performed according to Standard Methods for the Examination of Water and Wastewater, number 3030 B. Fisherbrand® 0.45 μm filters were used for the analysis.

All dissolved, sorbed, and total copper samples were analyzed on a Perkin-Elmer 703 flame atomic absorption (FAA), which had a range of 100 to 5000 μg/L. Samples measuring less than 100 μg/L were analyzed on a Perkin-Elmer Zeeman 5100 HGA 600 graphite furnace atomic absorption spectroscopy, which had a detection limit of 1 μg/L and a range of one to 40 μg/L. Dilutions were performed as necessary.

Quality control measures were taken in the collection and analysis of samples through blanks, spikes, and triplicate samples.

Data were analyzed using t-tests and analysis of variance using NCSS 97 (1997). Regression analyses were also performed using Microsoft® Excel 97 (1996). An alpha of 5% was used for all statistical tests. Where appropriate, the data were checked for equal variance and normality to ensure the requirements for the statistical analysis were met.

<u>Tomato Plants</u>

During this four month greenhouse-scale study, tomato plants were successfully raised from seedlings to mature, tomato fruit producing plants. The healthy, mature plants were about one meter in height, half a meter in width, and produced tomatoes until the plants were removed in late October 1998. No pest infestations affected the plants.

<u>Runoff and Groundwater</u>

A plot of average total suspended solids concentrations from the third through eighth simulated rainfall events is shown in Figure 2. A linear regression analysis performed on the average total suspended solids concentrations indicated no significant change in the runoff suspended solids concentration over time for any of the three simulated fields ($p > 0.15$ for each).

A one-way ANOVA was performed on the total suspended solids concentrations in the runoff from both simulated plasticulture fields and the control field. No significant difference in total suspended solids concentrations for the three field types was detected ($p = 0.244$). For the simulated plasticulture fields, the average suspended solids concentration was 1758 ± 1169 mg/L. The control field, which contained only natural soil cover, had an average suspended solids concentration of 1490 ± 996 mg/L.

Figure 3 shows a plot of the average total, dissolved, and sorbed copper concentrations in runoff samples collected during the third through eighth simulated rain events. Note that the samples from the simulated plasticulture field with sedimentation were collected from the runoff, prior to sedimentation control. Total copper concentrations in the plasticulture runoff samples range from 4040-4770 μg/L in the beginning to 1445-2070 μg/L at the end of the rainfall events. Dissolved copper concentrations ranged from 224-652 μg/L in the beginning to 155-286 μg/L at the end. In comparison, total and dissolved copper concentrations in the runoff from the control field ranged from six to 31 μg/L and 30 to 430 μg/L, respectively.

A regression analysis on the total, dissolved, and sorbed copper data from each of the simulated plasticulture fields indicated results similar to those for suspended solids. No significant trends over time were found for any forms of copper on either of the simulated plasticulture fields ($p > 0.22$).

A two-sample t-test was performed on the total, dissolved, and sorbed copper results from all of the runoff composite samples collected from each of the two simulated plasticulture fields. No significant difference between the means from the two simulated plasticulture fields for the total copper concentrations ($p = 0.89$), dissolved copper concentrations ($p = 0.13$), or sorbed copper content ($p = 0.51$) was found.

Means and standard deviations for the total, dissolved, and sorbed copper concentrations in the groundwater draining from the simulated plasticulture fields were 312 ± 198 μg/L, 229 ± 90 μg/L, and 921 ± 704 μg/g, respectively. Two-sample t-tests were performed on the total, dissolved, and sorbed copper results from the groundwater composite samples from the simulated plasticulture fields. No significant difference between the means from the two simulated plasticulture fields for the total copper concentrations ($p = 0.13$), dissolved copper ($p = 0.82$), or sorbed copper content ($p = 0.56$) was found. The mean values for the groundwater from the control field were 42.3 ± 44.4 μg/L total copper, 14.1 ± 5.7 μg/L dissolved copper, and 306.9 ± 309.7 μg copper/g soil. A one-way ANOVA analysis was performed on the total, dissolved, and sorbed copper results from the groundwater composite samples collected from the control field and both simulated plasticulture fields. The one-way ANOVA showed a significant difference between the control field and both plasticulture fields for the total copper concentration, dissolved copper concentration, and the sorbed copper content ($p < 0.05$). The groundwater from the control field was significantly different in copper concentrations because copper-based crop protectant was not applied to this simulated field.

The volumes of runoff and groundwater leaving each tomato field from the simulated rain events were recorded and a one-way ANOVA was performed to determine if the volumes differed significantly. The ANOVA analysis showed no significant difference between runoff volumes generated on either of the simulated plasticulture fields or the control field not using plasticulture ($p = 0.24$). The runoff leaving the tomato fields accounted for 10-13 liters of the applied 54 liters of rainfall. The ANOVA analysis showed no significant difference between groundwater volumes generated from the fields ($p = 0.054$). The groundwater generated ranged from 9 to 17 liters of the applied 54 liters of rainfall (Figure 4).

Distribution of the Copper-based Crop Protectant

For each rainfall event, a mass balance was performed on the plasticulture tomato fields to determine the distribution of the applied copper-based crop protectant. Results from the mass balances show that following a storm, approximately 99% of the applied copper remained on the agricultural fields both sorbed to the soil and on the leaves of the tomato plant.

Analysis of the soil cores showed that of the copper remaining on the soil, most was found within the top 2.5 centimeters of soil depth. For both simulated plasticulture fields, the elevated copper levels in the top 2.5 centimeters averaged 26 ± 18 mg copper/kg soil along the compacted region of soil between the plasticulture covered mounds. Figure 5 shows the location in the fields where soil cores were taken. Contour plots of the copper distribution within the plasticulture field soil are shown in Figures 6 and 7. Figure 6 is a plan view of the distribution of copper sorbed to the top 2.5 centimeters of soil in the simulated plasticulture field. Figure 7 is a cross-sectional view of the copper sorbed to the soil at two different locations within the bin.

The mean copper content of the soil in the control field remained constant along the field surface and depth, averaging 2.65 ± 0.25 mg copper/kg soil. This copper content was less than the baseline copper content, which averaged 4.25 ± 0.18 mg copper/kg soil, determined at the beginning of the study. Some possible reasons for the decrease in copper content include desorption of the copper into infiltrating rainwater and uptake of copper by the tomato plants.

Figure 8 shows that of the one percent of copper that did leave a tomato field, most of the copper (74%) was sorbed to the total suspended solids in the runoff. The remaining copper was partitioned as follows: eight percent as dissolved copper in the runoff, ten percent as dissolved copper in the groundwater, and eight percent sorbed to the fine soil particles escaping in the groundwater.

<u>Effectiveness of Sedimentation Control</u>

A soil settleability study, performed in Imhoff cones, on the runoff generated from the September 25, 1999 sampling event showed that the soil settles very rapidly upon collection into a settling basin (Figure 9). In fact, all possible settling had occurred within the first five hours.

Figure 10 shows a box plot of the change, from five days of settling, in the total and dissolved copper concentrations for samples from all of the rain events. Over the five-day settling period, median total copper concentrations decreased from an initial concentration of 2100 μg/L copper to a final concentration of 250 μg/L copper, a 90% reduction. Dissolved copper concentrations remained relatively stable throughout settling, with median initial and final concentrations about 150 and 140 μg/L copper, respectively.

Although five days was used as the settling period in this study, the most significant reductions in copper concentrations occurred within the first two days of settling (Figure 11). This is also the length of time in which the settling of suspended solids had stabilized.

A two-sample t-test was performed on the total and dissolved copper concentrations before and after sedimentation. A significant difference was detected in the total copper concentrations before and after sedimentation ($p < 0.01$); however, no significant difference was detected in the change in the dissolved copper concentrations ($p = 0.85$).

Although 99% of the applied copper remained on the tomato fields following rainfall, the results from this study indicate that the one-percent leaving the fields, in both groundwater and runoff, was sufficiently high to exceed known LC_{50} values for copper toxicity to aquatic organisms (LaBreche, 1998; Calabrese *et al.*, 1977). Mean concentrations of 189 ± 139 μg/L dissolved copper and 2102 ± 433 μg/L total copper were found in the runoff. This is consistent with the 20 to 250 μg/L dissolved copper concentrations found in actual agricultural runoff from full-scale tomato plasticulture fields (Brady *et al.*, 1999; Dietrich *et al.*, 1999). Mean concentrations in the groundwater from the simulated plasticulture fields were 312 ± 198 μg/L total copper and 216 ± 99 μg/L dissolved copper.

Past research has found that typically one to two percent of the applied pesticide mass leaves the field in runoff (Wauchope, 1996) and less than one-tenth to one percent leaches below the root zone, contributing to groundwater contamination (Flury, 1996). This is consistent with results found in this study, in which a total of one-percent applied copper was lost, with 0.82% in the runoff and 0.18% in the groundwater. Endosulfan, an insecticide, was also found to exhibit similar results with 0.2% of the applied mass leaving agricultural fields as runoff (Volger, 1997).

A regression analysis showed no significant difference in total suspended solids concentrations or dissolved, sorbed, and total copper concentrations throughout the 20-minute rainfall event, meaning a first flush was not observed. No significant difference in total suspended solids concentrations was detected in runoff from the simulated plasticulture fields and the control field with a natural soil cover. It is possible that the control field did experience the same level of compaction as a real agricultural field.

Analysis of soil cores taken from the tomato fields showed that most of the copper remaining on the fields was in the top 2.5 centimeters of soil. This is consistent with results for the pesticides aldrin, DDT, and lindane reported by Lichtenstein (1958). He reported that seventeen months after pesticide applications 84-96% of the remaining chemicals were found in the top 7.6 cm of soil, 4-12% were found from 7.6 to 15.2 cm, and 0-5% were found in 15.2 to 23 centimeters. Mobility of the individual pesticides did not affect their distribution in the soil. The lack of copper in the deeper soil could be explained by preferential flow pathways, directing the copper to the underdrain system which served as the groundwater table. It is also possible that the copper complexed with water soluble organic and inorganic chemicals or was retained on colloids. Grout *et al.* (1999) reported that sorption of copper to colloids provided a main means of metal transport for urban storm water runoff.

With most of the copper (74% of the one-percent loss) leaving the tomato fields sorbed to the soil suspended in the runoff, sedimentation proved quite effective in reducing the total copper concentration to receiving waters. A 90% reduction in the total copper concentration occurred over the five-day settling period used in this research. The most dramatic reduction in copper occurred during the first day of settling, with the concentration stabilizing the second day. However, the dissolved copper concentration was not affected by sedimentation, suggesting the added time for the soil-storm water interaction did not change the equilibrium. In other words, there was no further sorption of the dissolved copper to the soil throughout the settling period. The results from the sedimentation basin in this study were consistent with the 40-87% soil reduction and 30-40% reduction in sorbed nutrients reported by Novotny and Olem (1994). The results are also consistent with those found by Scott *et al.* in which a 90% reduction of endosulfan in estuarine water was reported from the use of a retention pond on a nearby farm.

Copper-based crop protectants are often applied to tomato plasticulture fields to reduce the incidence of bacterial and fungal diseases, which can affect crop yields. This farming practice has led to high copper concentrations in agricultural runoff that flows directly to adjacent waterways (Brady *et al.*, 1999; Dietrich *et al.*, 1999; Klawiter, 1998; Dietrich *et al.*, 1996). The concentrations of copper in the receiving waters have at times exceeded water quality standards and threatened the aquaculture industry and local aquatic resources.

The greenhouse-scale simulation, in which copper-based crop protectants were applied to simulated tomato fields, confirmed results reported in watershed-scale investigations on the Eastern Shore of Virginia (Brady *et al.*, 1999; Dietrich *et al.*, 1999; Klawiter, 1998; Dietrich *et al.*, 1996). High copper concentrations were reported to leave the simulated plasticulture fields as both total and dissolved copper in the runoff; the values were 2102 ± 433 μg/L and 189 ± 139 μg/L copper, respectively. In the groundwater from the simulated plasticulture fields, the total copper concentrations averaged 312 ± 198 μg/L and the dissolved copper concentrations averaged 216 ± 99 μg/L.

Similar to results from previous pesticide research, 99% of the applied copper was found to remain on the agricultural fields. Most of the copper sorbed to the soil was found in the top 2.5 centimeters of soil. With substantial amounts of copper sorbed to the field soil, it was not surprising that most of the copper in the runoff was found to be sorbed to the suspended soil; consequently, sedimentation control proved highly effective in reducing overall copper loadings to the receiving waters. On average, there was a 90% reduction in the total copper concentration from settling. However, the dissolved copper concentration remained relatively stable at about 250 μg/L copper throughout the settling process, suggesting the retained water could still pose a threat to aquatic life.

Based on the high dissolved copper concentrations remaining after settling, sedimentation control cannot be accepted as an effective best management practice in all situations. The effectiveness of sedimentation control will depend on characteristics of the agricultural field including the grade of the land, proximity to waterways, placement of the sedimentation basin on the field, and ability of the receiving water body to adequately dilute the runoff.

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