Modeling the Costs and Environmental Benefits of Disposal Options for End-of-Life Electronic Equipment: The Case of Used Computer Monitors

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Abstract

Managing the growing quantity of used electronic equipment poses challenges for waste management officials. In this paper, we focus on a large component of the electronic waste stream—computer monitors—and the disposal concerns associated with the lead embodied in cathode ray tubes (CRTs) used in most monitors. We develop a policy simulation model of consumers' disposal options based on the costs of these options and their associated environmental impacts.

For the stock of monitors disposed of in the United States in 1998, our preliminary findings suggest that bans on some disposal options would increase disposal costs from about $1 per monitor to between $3 and $20 per monitor. Policies to promote a modest amount of recycling of monitor parts, including lead, can be less expensive. In both cases, the costs of the policies exceed the value of the avoided health effects of CRT disposal.

Key Words: end-of-life electronics; waste stream; cost-benefit analysis

JEL Classification Numbers: Q2, Q0, H8

Contents

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I. Introduction

The growing importance of information technology to the world economy and to consumers in the United States and other developed countries has brought about a surge in demand for electronic equipment. For example, according to recent estimates, shipments of personal computers in the United States grew from slightly more than 10 million units in 1992 to just more than 30 million units in 1997 (National Safety Council 1999). At the same time, the rapid pace of advances in computing technology often means the useful life of electronic equipment grows shorter and shorter with each successive generation. For example, in 1997, the average life span of a computer tower was four to six years; by 2005, it is expected to be just two years (Salkever 1999). As a result, a growing fraction of the increasing stock of many types of existing electronic equipment becomes obsolete each year.

Managing the growing quantities of used electronic equipment poses challenges for waste management officials. A primary concern is that such equipment can contain hazardous materials, such as heavy metals or lead, that could be released into the environment during incineration or concentrated and then dispersed in incineration ash. For example, most computer monitors and color televisions use cathode ray tubes (CRTs) containing lead to shield users from radiation. This lead could pose an environmental

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hazard when used CRTs are incinerated. In the United States alone, some experts say that approximately one billion pounds of lead from computers and other electronic equipment will enter the waste stream within the next decade (Salkever 1999).

Dealing with used electronic equipment also is a challenge for the businesses and households that generate the waste. In the United States, under Subtitle C of the Resource Conservation and Recovery Act, large commercial and industrial generators of used CRTs must dispose of that equipment at a hazardous waste facility, which can be even more costly than disposal by small commercial generators. With the exception of some jurisdictions that now restrict all landfilling and incineration of CRTs, small commercial generators and households are exempt from this requirement and can dispose of used CRTs with the rest of their household trash. However, many community waste disposal programs are reluctant to pick up this equipment at the curb. In practice, both large and small businesses and households appear to be storing their used CRTs.[1] In recent years, a growing number of large quantity generators of all types of electronic equipment are finding it economical to send their used equipment to recycling facilities, but this practice is not widespread (National Safety Council 1999).

Several policy proposals have been put forth to increase the recycling of general electronic equipment, as well as specific types of electronic equipment such as CRTs. Since April 2000, Massachusetts has banned disposal of CRTs at all municipal solid waste (MSW) landfills and incinerators. This ban is being coupled with the establishment of several CRT drop-off sites throughout the state, as well as other efforts on behalf of the state to promote use of these facilities and other means of CRT recycling. Other communities, in California, New York, Minnesota, New Jersey, Virginia, and Illinois have experimented with various types of collection programs, including one-day drop-off opportunities for consumers to bring in old equipment; the siting of permanent depots for disposal of equipment; curbside collection; and point of purchase (retail) collection. Some approaches also seek to give manufacturers the responsibility for funding the disposal or recycling of machines that they have produced. Manufacturers have opposed these approaches, noting that they already participate in and underwrite many pilot projects to reclaim old computers.[2] Some manufacturers also have instituted

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[1] A 1995 Tufts University study (Eustace, et al. 1995) estimates that 75% of end-of-life CRTs are in storage.

[2] U.S. Environmental Protection Agency (1998, 1999) discusses some of these programs in detail.

programs under which consumers may return used equipment for a fee, and the manufacturers donate useable equipment to charity and dismantle the rest.

In April 2001, the European Commission (E.C.) submitted the "Directive on Waste from Electrical and Electronic Equipment" to the European Parliament. Among its many articles, this proposal calls on European Union member states to require distributors and manufacturers to take back electrical and electronic equipment and sets ambitious recycling rate goals for that equipment. The E.C. proposal also encourages member states to establish minimum recycled content standards for new electronic products. Several countries, including the U.K., Belgium, Sweden, the Netherlands, and Japan, are developing their own regulations to establish take-back systems for electronics. A prominent related issue is the effect of take-back requirements on overseas companies that manufacture and export electronics.

In this project, we develop a model of consumers' options for discarding computer monitors at the end of the equipment's useful life (end-of-life or EOL). We use this model to analyze the effect of different policy options on behavior of different types of monitor consumers (specifically, several categories of residential and nonresidential consumers), on the cost of achieving a particular recycling target or a particular amount of reduced disposal, and on the environmental consequences. As we discuss below in our description of environmental effects, the environmental consequence that we focus on is the effect of lead releases from CRT incineration on human health.[3]

The paper is organized as follows. First, we provide a brief overview of the composition of a CRT and the environmental concerns created by CRT disposal and throughout the CRT lifecycle. Next, we provide a summary of public policies and other programs that have been adopted in the United States and elsewhere to promote CRT recycling. In the next two sections, we describe our approach to modeling CRT disposal decisions. Here, we specify the consumer's purchase and scrappage decisions and our simulation model, including its environmental damages module. We also discuss some limitations of the modeling approach. We then present the results of our baseline and policy simulations, and conclude with a discussion of our findings and directions for future research.

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[3] According to estimates by McKenna et al. (1996) lead emissions into the atmosphere from incineration of used CRTs are greater per ton of CRTs disposed than are potential emissions into landfill leachate.

II. An Overview of CRTs

Found mainly in television sets and computer monitors, CRTs are one of the most common components of discarded electronics. Stanford Resources has reported that the worldwide market for CRT monitors was 84.2 million units in 1997 and it is expected to grow to 100 million units in 2002 (Stanford Resources Inc. 2000). The rapid development of new and technologically superior CRTs and other display options is creating an abundance of obsolete equipment in the waste stream. For example, the National Safety Council (1999) estimates that 16 million computer monitors became obsolete in the United States in 1998. Once a CRT becomes obsolete, it must eventually be discarded. Waste management officials view the potentially hazardous components found in these discarded CRTs to be a growing challenge.

In this project, we focus on CRT displays found in computer monitors. A CRT display is typically composed of a glass panel, a cathode ray tube, a casing, connecting wiring, and shielding. Figure 1 illustrates a computer monitor, denoting those parts that are plastic or steel, and showing the predominantly glass CRT component.

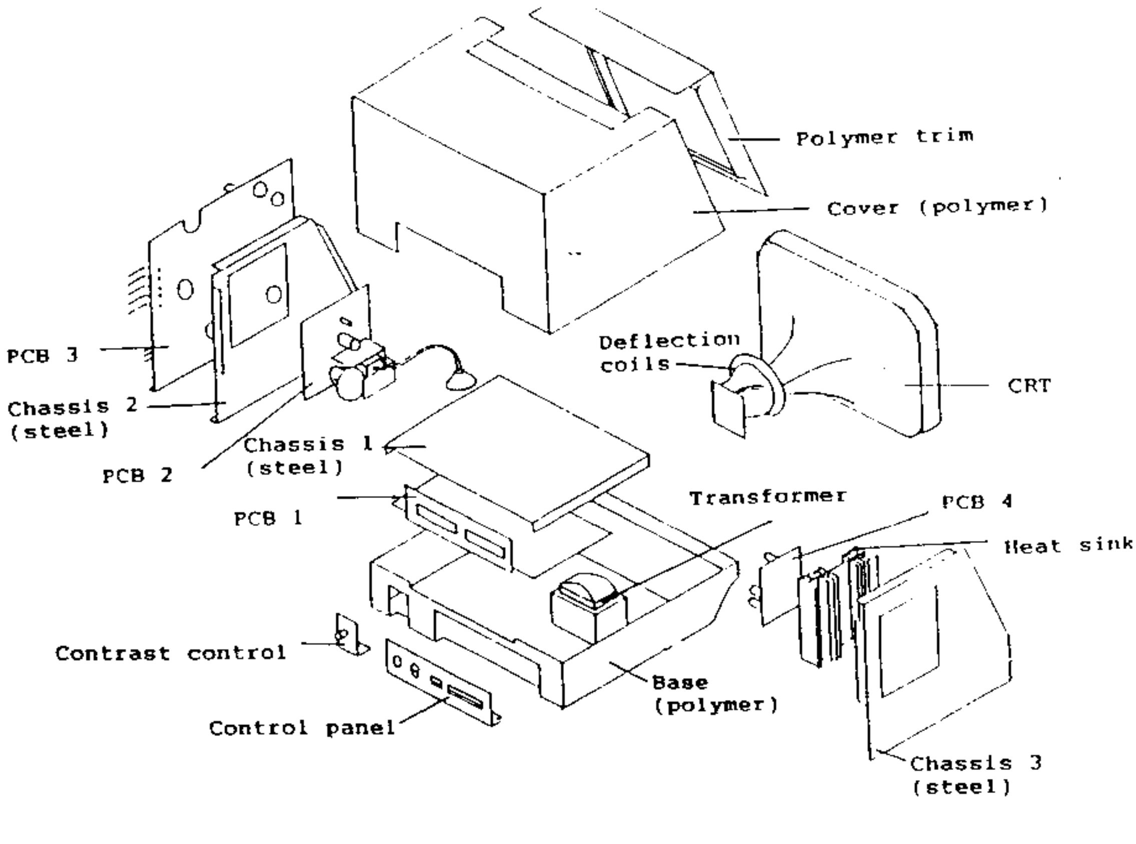


Figure 1: Components of a Computer Monitor

Figure 2 is a schematic of the CRT itself. CRTs are comprised mostly of glass with significant amounts of lead and smaller quantities of cadmium and other metals. Lead in the glass of the cathode ray tube itself is the major source of lead in the display. This lead constitutes the major use of lead in the entire material composition of a typical desktop computer system (computer and monitor); to a much lesser extent, some lead is used in the soldering of circuit boards and other components.

Leaded glass in the CRT is found mainly in the neck, funnel, frit, and, to a smaller extent, the face panel. The frit seal, used only in color CRTS, binds the funnel to the face glass and forms an envelope in which screen materials are sealed (FCSHWM 1999). The seal serves as shielding to separate electronic beams in the color spectrum. Table 1 shows estimates of the quantities of lead found in the different types of glass used in both a color and monochrome CRT.

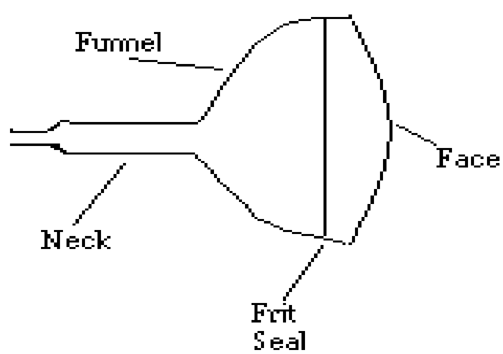


Figure 2: Schematic of a CRT

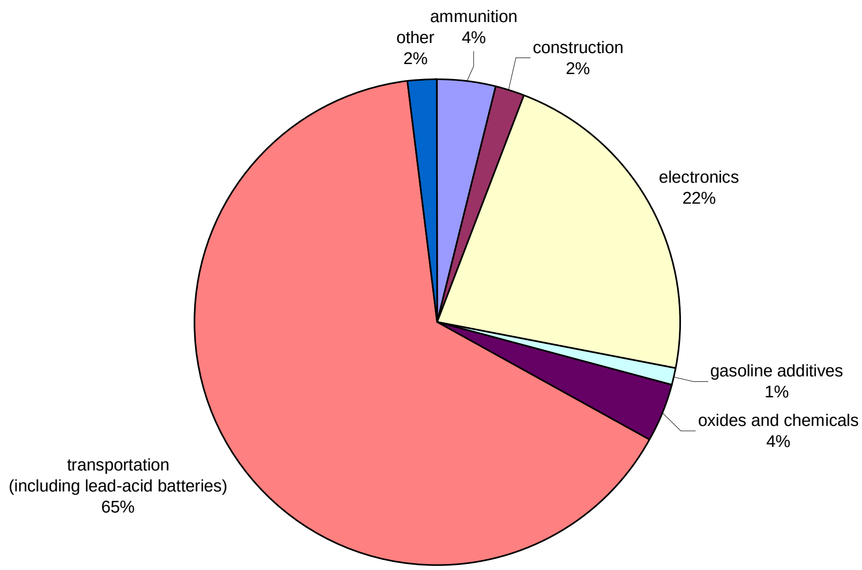
Table 1: Lead Content in CRT Glass Components by Mass

Glass	Color CRT	Monochrome CRT
Panel (Face)	0% - 3%	0% - 3%
Funnel	24%	4%
Neck	30%	30%
Frit	70%	N/A

Source: Townsend et al. (1999).

A. Environmental Concerns

Concerns about environmental and health effects from lead entering the environment have been a major reason for the increased efforts to reduce the number of CRTs in the waste stream. Lead is used in several products in the United States (Figure 3), including lead-acid batteries and other transportation products (65% of total lead used by weight) as well as CRTs and other electronic products (22%). While CRTs and other electronics do not account for the primary use of lead, they comprise the largest proportion by weight of lead entering the solid waste stream (Figure 4). This proportion may be partially explained by restrictions on disposal of lead-acid batteries in MSW landfills and increased battery recycling. As of 1991, 35 states had banned the disposal of batteries in landfills (EPA 1991). In addition, the role of secondary lead is key in the lead market. Recycled lead makes up 76% of the total refined lead produced in 1999 (Smith 1999).



Source: Matthews et al., 2000

Figure 3: Uses of Lead in the United States (1996; percent by weight in 1000 metric tons)

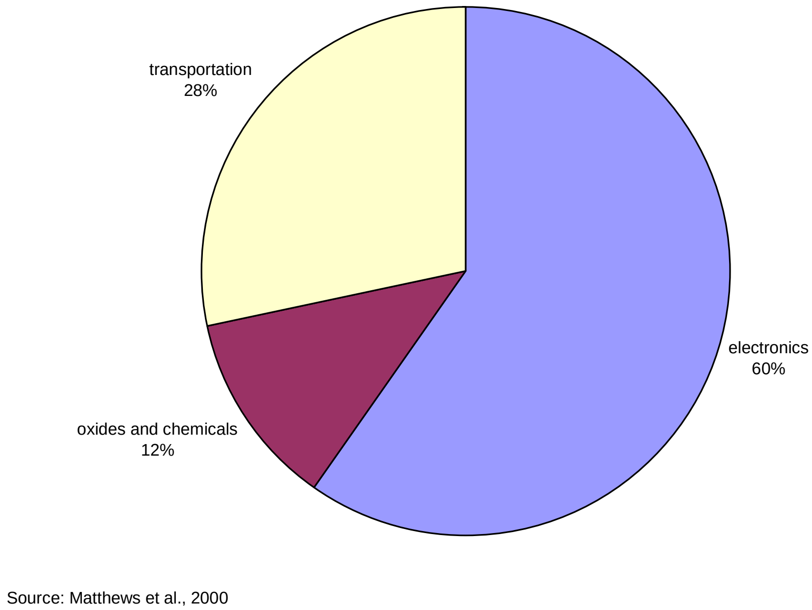


Figure 4: Lead in the Solid Waste Stream (1996; percent by weight in 1000 metric tons)

Our focus in this paper is limited to the costs of environmental and health effects associated with disposal of CRTs contained in computer monitors. However, it is important to note that environmental releases of lead and other hazardous substances can take place throughout the monitor's life cycle.[4] For example, the extraction and processing of the raw materials used in CRT production—as well as the fabrication of the CRT—may lead to environmental releases of lead and adverse health effects. The mining and manufacture of lead used in CRTs results in emissions of lead into the environment: lead mining results in solid by-products released into the environment, while lead smelting and the production of lead oxide (the form of lead used in CRT glass)

[4] See Caudill et al. (2000) for additional discussion of the lifecycle impacts of CRTs.

result in lead emissions into the air as well as solid by-products that contain lead, which are subsequently disposed and thereby disseminated.[5]

The CRT production process itself may involve air and water emissions of pollutants including solvents and their vapors, acids, chelating agents, surfactants, caustics, and glass wastes.[6] Computer manufacturing workers also are exposed to these substances.

The process of CRT end-of-life management also could result in effects on human health and the environment in addition to effects associated with lead-containing components. Plastic from computer equipment is supposed to be much more suitable for further processing than plastic from televisions and other products because it is more homogenous in its chemical composition. However, the extent to which incineration may result in emissions of flurons and possibly dioxins if the plastics contain halogenated substances is controversial among researchers. In addition, at present, the cost of recycling exceeds the market value of the plastic. Many experts also agree that plastics are safe in waste-to-energy incineration. On the one hand, then, CRT recycling may result in energy savings when plastic, glass, and steel are reused. On the other hand, if the CRT must be transported a lengthy distance to a recycling center, the transportation energy used may be greater than the amount of energy saved through recycling.

In the context of our model, the disposal policies we address may indeed affect these various stages of the lifecycle. As we noted above, however, in the United States at least, lead used in electronics is not the major use of processed lead—that is, lead from either mining (primary) or secondary sources. To some extent, end–of–life policies for monitors would influence the extraction and processing stages only insofar as recycling, for instance, reduces demand for primary lead (although secondary lead smelting also has environmental effects). Other stages depend critically on a host of parameters outside the scope of our model, such as the effectiveness and cost of enforcing occupational safety and health provisions; other inputs that might be substituted for the plastics and coatings in monitors; and the environmental effects of the recycling processes themselves. We

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[5] Perwak et al. (1981) provide detailed information on the distribution of lead in the environment and its effect on humans and other organisms.

[6] For a more technical discussion of the CRT production process, see Mizuki and Shuldt (2000).

hope that our model can be a starting point for research extensions to address these issues.

B. Liability Issues

In the United States, nonresidential consumers using large quantities of monitors are subject to federal RCRA regulations for hazardous wastes because the equipment often fails to pass the Toxicity Characteristic Leachate Procedure (TCLP) test for lead. The TCLP test involves the crushing of the substance in question and testing to determine the leachability of the materials. It is intended to simulate a 20-year decomposition process in a landfill (Biddle 2000). CRTs that have lead concentrations greater than 0.05 mg/L fail the test and are considered hazardous. Color CRTs almost always fail this test while monochrome CRTs usually pass and do not have to be treated as hazardous waste (Townsend 1999). Businesses that use small quantities of CRTs and residential CRT owners also are exempt from these RCRA requirements.

In addition to RCRA, commercial and industrial generators of electronic waste also must be concerned with the liability provisions in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). This act addresses the release of hazardous materials and the future remediation of toxic waste sites. The desire to avoid future liability has led many commercial organizations to develop alternative methods—such as manufacturer take-back or leasing agreements—for disposing of their electronic equipment.

Although Figure 4 suggests the contribution of CRTs to lead found in the solid waste stream is large, the distribution among local waste facilities is a topic of much debate. Some states and localities, such as New Jersey and Hennepin County, MN, claim that high levels of lead and cadmium in their municipal waste can be attributed to CRTs. Industry experts dispute these assertions and claim that municipal landfills do not have lead in their leachate as a result of CRTs. Nonetheless, concerns about lead in the waste stream and the environmental hazards or potential liabilities associated with that lead have provided an impetus to both public and private efforts to reduce disposal and increase recycling of CRTs. Towards further discussion of these and other policy concerns, in the next section we review a number of different policies and programs that have been adopted or proposed to promote recycling of used CRTs.

III. Policies and Programs to Facilitate and Promote Recycling of Used CRTs

There are several examples of policies and programs to promote recovery and recycling of used CRTs in the United States and in other countries. Until recently, privately initiated efforts to promote CRT recovery (and recovery of other electronic equipment) have been limited and directed at nonresidential CRT users. These programs can include leasing programs, warranty programs, and explicit take back agreements. In the past year, new programs addressing the residential market have been announced by private companies such as IBM. The response to these programs is not yet clear. Aside from these initiatives, recovery of CRTs from residential users is, at present, generally the result of explicit public policy decisions. In this section, we offer an overview of some of these approaches. In Appendix A we offer more details drawing on information from the U.S. Environmental Protection Agency.

A. Private programs

Most private programs are based on the relationship between the retailer and the customer and include leasing, warranties, trade-ins, repair services, and take-back of equipment by retailers or manufacturers. These practices are generally limited to high-value electronic items such as computers, copiers, printers, and communications equipment (MOEA 1995). Several companies have announced programs for taking back electronics. For example, in November, 2000, IBM announced a recycling program for individual consumers and small businesses in which any manufacturer's computers and computer-related equipment can be sent to IBM for $29.99 per specified volume of the shipping container (26" x 26" x 26"). IBM donates useable equipment to charity and demanufactures nonuseable equipment.

It is too early to ascertain the success of approaches such as that taken by IBM. Generally, the waste generated by nonresidential consumers of electronic products is typically more homogenous (i.e. large quantities of similar brands and vintages and, therefore, similar components and construction) than that generated by households, and therefore more valuable to demanufacturers. In addition, primarily for nonresidential consumers, leasing has become increasingly popular, especially with information technology equipment, as a means of ensuring that a company is not stuck with obsolete products that must be placed in storage or disposed. Leasing agreements are often structured in such a way that any potential liability associated with the products shifts from the commercial consumer to the retailer or manufacturer.

B. Local Programs

Table 2 provides examples of various methods currently employed by U.S. localities in the collection of waste electronic products generated by individuals and small businesses. In many cases these approaches are being tested as pilot projects.

Table 2: Summary of Residential Collection Methods

Collection Method	Description	Barriers	Advantages
Drop-off Event	One– or two–day event typically held at existing municipal facility	Significant publicity is needed to encourage participation	Short time frame and potential for high collection amount
Regional Collection	Similar to drop-off event with the exception that multiple communities are involved	Potential for unequal distribution of costs across communities	Large base of residents; economies of scale over single community event
Curbside Collection	Collection of electronic waste is done periodically or by request	Operating costs can be high	Low participation costs for households
Permanent Collection Depot	A permanent collection location for waste electronics is opened at an existing collection site for other waste items	Not effective for small communities; staffing needs raise operating costs	Convenient, year round collection of equipment; economies of scale possible
Point of Purchase Collection	The retailer covers the cost of collecting and storing electronic waste	Requires active participation of retailer	Low operating costs for municipalities; retailers can easily promote the program

Source: Adapted from "Analysis of Five Community Consumer/Residential Collections—End of Life Electronic and Electrical Equipment." EPA, Common Sense Initiative, April 1999.

One local program, in particular, is illustrative of these approaches. In 1992, Hennepin County Minnesota, which includes the city of Minneapolis, began collecting electronic equipment from households. Initially the program was designed to get heavy metals contained in electronics out of the municipal waste stream. Several collection methods are used, including permanent drop-off sites and periodic collection at the curb. In 1997, Minneapolis added regular curbside collection within the city limits. In 1999, the program collected approximately 43,000 electronic products weighing, in aggregate, approximately 850 tons. CRTs collected in this program are currently sent to a secondary lead smelter (American Plastics Council 2000).

C. State Policies and Programs

Several states also have adopted policies to promote CRT recycling. In Florida, CRTs are designated as hazardous waste only if they are landfilled or incinerated. If CRTs are reused to make new CRTs or other commercial products, they do not have to be treated as hazardous waste. The state also is providing financial assistance and grants to local governments to help develop the infrastructure necessary to make recycling electronics cost-effective (Clarke 1999). In Massachusetts, the Department of Environmental Protection banned the disposal of CRTs from its landfills and incinerators beginning in April 2000. In January 2001, the state also received authorization from EPA to exempt intact CRTs destined for reuse or recycling from hazardous waste regulations (see *Federal Register* 15 November, pp. 68915-68919).

D. Federal Policies and Programs

At the federal level of government, there are few regulations in place to promote CRT recycling within the United States. The only exceptions are the restrictions and liability risks for commercial users under the RCRA and CERCLA regulations (see earlier discussion) and the Taxpayers Relief Act of 1997, which provides tax incentives for companies that donate technology equipment to schools (Lightly 2000). Also, the federal government has become involved in voluntary programs and local initiatives to curb electronic waste. The U.S. federal government, as well as many state governments, are most concerned with developing markets for recycled products prior to instituting any legislation requiring mandatory take-back programs (South Carolina State Senate 1999).

The federal government also is making some effort to remove existing barriers to CRT recycling. In 1998, the Computer and Electronics Sector Subcommittee of EPA's Common Sense Initiative Council recommended to the EPA that it should remove unnecessary regulatory barriers to CRT glass-to-glass recycling, specifically recommending that CRTs destined for this end-use should be exempt from hazardous waste management requirements under RCRA. The EPA is working on a proposed rule to implement this recommendation.

The EPA also is working with industry groups to promote Design for the Environment (DfE) initiatives. These initiatives seek to limit adverse impacts on the environment throughout a product's lifecycle, including manufacturing, product, use and disposal. DfE initiatives for electronics promote pollution prevention in the design and manufacturing process and the development of computers that use less energy. To address concerns about product disposal, DfE initiatives promote making products that are more easily recyclable and increasing the use of recycling materials in the manufacturing processes (Biddle 2000). Partly in response to these efforts, both IBM and Apple have developed computer systems that are easier to upgrade and recycle.

E. The EU Directives

On the international front, there also are several policy initiatives, particularly across Europe. Although many member states of the European Union already have drafted legislation related to this waste source, the European Environment Council has determined that different national policies could detract from the effectiveness of recycling policies. For this reason, two proposed community-wide directives have been issued.

The first—The Directive on Waste Electrical and Electronic Equipment—aims to reduce the generation of and encourage the reuse and recycling of electronic waste. This directive is based on Article 175 of the treaty establishing the European Community that states that community policy on environmental issues should strive for a high level of protection while taking into consideration the great diversity within the community (EC 2000). The directive requires that 60%–80% of electronic equipment be recovered and recycled by manufacturers by 2006 (Hileman 2000).

The second—The Directive on the Restriction of the Use of Certain Hazardous Substances in Electrical and Electronic Equipment—aims to minimize the risks and environmental impact of the treatment and disposal of electronic waste. It is based on Article 95 of the E.C. treaty. This second directive will phase out the use of hazardous substances—lead, mercury, cadmium, hexavalent chromium, polybrominated biphenyls (PBBs) and poly brominated diphenyl ethers (PBDEs)—in electronic products by 2008 (Hileman 2000).

The international response to this initiative has been mixed. Trade groups both within Europe and the United States claim the cost to industry will be unreasonably high (Recycling Laws International 1999). American companies are using agreements made under the World Trade Organization (WTO) as a basis for claiming that these directives are an illegal barrier to trade and this issue was a major point of contention during the WTO meetings in Seattle in 1999 (Biddle 2000).

The European Council of Ministers and the European Parliament must approve the directive proposals before they can go into effect. Several issues need to be addressed before they will be approved. For example, the current agreements between members of the EU must be modified to address the regulations in the new directives. Once in place, the directives would apply to all manufacturers of electronic equipment that sell their products to European consumers.

F. Japan's Initiatives

In Japan, policies for disposal of electronic equipment are under active development. In January 2001, the Ministry of the Environment published a draft proposal to require a recycling rate of 55% for CRT monitors. Japan's existing waste management policies are generally grouped under the "Legislative System for Promoting the Creation of a Recycle-Oriented Society " (Ministry of International Trade and Industry 1998), which establishes a variety of recycling and take-back mandates. In a January, 2000 report, the Clean Japan Center, a semi-governmental organization under the Ministry of International Trade and Industry, argued for the "Law for Recycling of Specified Kinds of Home Appliances" to be extended to cover electronic and related equipment discarded by residential consumers (see "Recycle-Oriented Society: Towards Sustainable Development," Clean Japan Center). Under this law, a take-back program for used home appliances (TVs, refrigerators, air conditioners, and washing machines) requires retailers to take back the products and transfer them to original manufacturers and importers; municipal offices are permitted to recycle office appliances (such as air conditioners) themselves. Manufacturers and importers are allowed to charge recycling fees to retailers; the fees “must not be above the appropriate costs for efficient recycling, and should be set at an appropriate level [when passed on to the customers] so as not to discourage discard by consumers.” The Ministers of Welfare and Trade are to survey retailers, manufacturers, and importers to ensure their compliance. Other laws under the legislative system address recycling of containers, packaging, and construction materials; “recycling” includes reusing products; removing parts and reusing them; and removing parts and materials and reusing them as fuel.

IV. Our Approach

The programs and policies described above suggest the usefulness of analysis to compare the effectiveness of alternative approaches. The goal of our modeling exercise is to determine the cost and environmental consequences of various methods of managing monitors at the end of their lives.

Our approach consists of two steps. First we describe the purchase and scrappage decisions of consumers. We borrow from three strands of literature in this step: capital stock optimization; scrappage functions for durable goods; and models of the private and social costs of waste disposal. In the second step, we construct a framework for the CRT product lifecycle and estimate a simulation model for the disposal stage of this lifecycle. Our model, the Computer Monitor Policy Simulation, or COMPS, tracks what happens to residential and nonresidential (business and government-owned) monitors in the United States once they are retired at the end of their useful lives. We use U.S. data to estimate CRT retirements for 1998 and then assess how many of those CRTs will be discarded according to a variety of discard options. For CRTs that are sent to an incinerator, the model also tracks the air emissions of lead, the potential health effects of those emissions and the associated damages in monetary terms. We then evaluate the private and social costs associated with policy options to influence choice of discard method.

A. The Consumer's Purchase and End-of-Life Scrappage Decision

The end-of-life discard decision upon which we focus is part of the larger decision that both residential and nonresidential consumers face in using computer services. This decision involves purchasing a monitor, typically for use as a complementary good with the computer system, and then discarding the monitor once it has become technologically obsolete, has physically depreciated and may no longer be workable, or has reached some combination of obsolescence and wear-and-tear.

In this section, we discuss the purchase and discard decisions. The purchase decision is key for two reasons. First, two parameters in the purchase decision bear directly on the discard decision: the rate of wear and tear, and the rate of technological obsolescence. Second, it may well be the case that policy options to influence monitor discard decisions operate on the purchase decision as well (for example, a deposit-refund policy that requires a deposit upon purchase and a refund upon return).

The purchase decision

We follow convention in standard vintage capital models of the purchase of durable goods. We assume that purchases of durable equipment are based on profit maximization in the context of a production function (for firms, and in the context of household production function for consumers).[7] Thus, firms follow expression (1) in combining computing equipment purchased at time v, $I(v)$, with L_v and K_v, labor and other capital of vintage v purchased at time t, according to

$$Q_v(t) = A(t)L_v(t)^{\alpha(t)}K_v(t)^{\beta(t)}\{I(v)e^{\gamma v}\, e^{-\delta(t-v)}\}^{1-\alpha(t)-\beta(t)} \qquad (1)$$

In (1), $A(t)$ is disembodied technological change; α and β are the usual Cobb-Douglas parameters; γ is the rate at which embodied technological change takes place in new computing equipment; and δ is the rate of physical decay. The rate of economic depreciation in these models is thus the sum of the rates of embodied technological change and physical decay.

[7] We follow Whelan (2000) in this discussion; his model is based on Solow's (1959) vintage capital model.

Under profit maximization, the cost of equipment discard arises as a cost in addition to other input costs:

$$\pi_v(t) = A(t)L_v(t)^{\alpha(t)}\, K_v(t)^{\,\beta(t)}\, \{I(v)e^{\gamma v}\, e^{-\delta(t-v)}\}^{1-\alpha(t)-\beta(t)}$$

$$- r_v(t)I(v)e^{-\delta(t-v)} - r^o(t)K_v(t) - w(t)L_v(t) - c_v^{\,i}\{I(v)e^{-\delta(t-v)}\} \qquad (2)$$

where the price of computers (without adjusting for the value of embodied features) of vintage v implies an annual rental rate of $r_v(t)$, the annual rental price of other capital is $r^o(t)$,and the wage rate is $w(t)$. The cost of discard for computer equipment of vintage v is $c_v^{\,i}$, where i denotes different discard options. In expression (2) we allow the cost of discard to vary with the vintage; for instance, younger monitors may be recyclable for parts whereas older monitors may be recyclable only for materials.[8]

Among the first order conditions, and of interest to us, is the condition describing the service value of a monitor at a point in time (formally, it is the standard Jorgensonian rental rate).[9] Using this gives the following expression

$$r_v(t) = \quad r_t(t) + \ c_v^{\,i} \qquad (3)$$

where r_t is the rental rate for a new monitor, and c is the cost of discard. When $r_v(t)$ exceeds $r_t + c_v^{\,i}$, the consumer could purchase (or rent the services of) a new monitor (and dispose of the old one) and derive services from the new monitor at lower rental cost than from the old monitor. Old monitors may be technologically inferior and

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[8] Discard options for younger monitors, in particular, may also include reuse (giving them to children in the family; donating them to charity). However, as we note later in the text, we do not include a reuse option in our model.

[9] As these conditions are routinely obtained, we don't reproduce them here (but see Whelan for their derivation).

also suffer wear and tear. These effects are captured in the price equation (4), in which the price of a monitor of vintage v_v at time t is

$$p_v(t) = \int_t^{\infty} r_v(s)\, e^{-(r+\delta)(s-t)}\, ds = p_t\, e^{gt}\, e^{-(\gamma+\delta)(t-v)} \tag{4}$$

and where the prices of new monitors change at rate g. Thus, monitors decline in price as they age not only due to general wear-and-tear but also because they become technologically obsolete as new and improved technologies become available.[10] For simplicity, and because anecdotal information supports our simplification, we assume that consumers make no expenditures on monitor repair during the lifetime of the monitor. Thus, our interpretation of "general wear and tear" is typified by gradual pixel degradation, for instance, that still allows the monitor to be used without maintenance expenditure.

The discard decision

Expression (3) is the basis of models of the consumers' scrappage decision for durable goods (for instance, see Hamilton and Macauley, 1999; Parks, 1977, and Alberini, Harrington, and McConnell, 1998). Our model explicitly incorporates technological change as a parameter governing economic depreciation, whereas the models of other durable goods, such as automobiles, generally do not include this term. The second extension we make to the standard scrappage model is the addition of disposal costs. From (3), if disposal costs are positive, they delay the discard decision essentially by increasing the rental service cost of switching to a new monitor.

In implementing our computer simulation model, we use expression (4) to test the reasonableness of the survey data that give us our starting point: the number of monitors discarded at the end of their lives. In particular, we have the survey estimate of t-v as six years, the approximate age at which monitors are reported by the survey to be

[10] In fact, Whelan (2000) has estimated that the National Income and Product Accounts seriously misstate the nation's physical stock of computers and computer related equipment by failing to account for technological obsolescence. These durables may still be productive even though they may be retired because they are no longer near the technological frontier. It is interesting to note that the effect of δ is increasingly significant in estimates made for the U.S. National Income and Product Accounts of the stock of productive computer goods.

discarded by their owners. Separately, we collected data on monitor prices to estimate g[11] and used depreciation guidelines for computers and related equipment used by the U.S. government (by the Department of Commerce and by the Internal Revenue Service). The survey-reported value performs well. (see Appendix B for details).

B. The Simulation Model

In order to analyze the private and social costs of different waste CRT management strategies and policy options, we construct a simulation model of the management of obsolete monitors by their owners. The Computer Monitor Policy Simulation Model (COMPS) tracks what happens to residential and nonresidential monitors in the U.S. once they are retired at the end of their useful lives. We use our national U.S. estimates of historic CRT sales to estimate recent CRT retirements based on the scrappage function and survey information, as we described above. The COMPS model then allocates retired CRTs across the different end-of-life discard options based on a cost-minimization algorithm that explicitly accounts for the heterogeneity of the costs associated with each of the different options for different classes of CRT users. We make the assumption that both residential and nonresidential consumers pick the least cost discard option among the options available to them. After obtaining our "baseline" results under these assumptions, we then exercise the model under several policy scenarios.

Our set of end-of-life (EOL) discard options for monitor management are based on our reading of the trade literature and the options represented in the different policies and programs we discuss later in our report. The options we include are:

Storage

Incineration

Municipal solid waste landfill

Drop off center recycling (residential consumers only)

Hazardous waste process (nonresidential consumers only)

[11] We collected price data on monitors during 1992 to 1999 from new monitors advertised in *The New York Times* business section during this period and chosen to represent monitors of comparable quality in diameter and color-capability.

Commercial recycling

For both residential and nonresidential consumers, we include storage as an end-of-life discard option because many consumers treat storage as a long-term disposal option.[12] In implementation of our model, we also assume that residential consumers themselves do not choose between incineration and landfilling but that instead, the waste haulers in their communities make this choice. For residential consumers, we also include the choice of taking used monitors to a drop-off center for recycling. For nonresidential consumers who generate large quantities of monitors for disposal, U.S. federal law requires that discards be treated as hazardous waste. Smaller nonresidential consumers are not subject to this legislation; while they may not engage in drop-off center recycling, they may use commercial recycling. In specifying this set of discard options, we assume that EOL monitors are not usable—for this reason, we do not include resale or charitable donation as options, as these alternatives usually are available only for monitors in working condition.[13] It also is important to note that our definition of "recycling" follows the conventional literature about monitor disposal in interpreting this as the de-manufacturing of the monitor for reuse of its materials, not "recycling" in the sense of reuse to reallocate a working monitor from one consumer to another. We also do not include a producer take-back option and we reserve the alternative of curbside-pick up for recycling to be evaluated as a policy option. We plan to consider the take-back approach in future research.

The simulation model seeks to predict the discard behavior of the population of heterogeneous monitor consumers and how their behavior responds to various policy interventions. The heterogeneity across consumers is attributable to differences in access to various EOL options and differences in costs for particular options. As an example of the former, in the United States, nonresidential establishments that generate a lot of monitor waste (more than approximately 85 monitors per year) must treat used monitors as hazardous waste and dispose of them in a hazardous waste facility, or send the CRT

[12] To be sure, monitors that are stored today, presumably, are eventually disposed by another of these options, but many consumers store indefinitely and the "snapshot" of discard behavior that we model thus includes storage as a discard option at "a point in time."

[13] Some nonworking monitors for resale may be sold for parts; we do not have estimates of how large this quantity may be.

for special hazardous waste treatment before disposing of it in a regular landfill.[14] These methods of disposal are substantially more expensive than simple disposal in a MSW landfill, an option that is generally available to all other classes of consumers.[15] As an example of the latter differences, storage costs also vary across the two consumer classes (residential and nonresidential) and within any particular class reflecting differences in types of dwelling spaces (storage in detached homes is assumed to be free while storage in apartments is not) and in differences in the rental cost of real estate.

To capture differences in storage costs and access to different discard options, we divide the population of used monitor consumers into six groups:

Residential consumers living in apartments and facing pricing of their waste collection (unit pricing)

Residential consumers living in apartments without unit pricing

Residential consumers living in houses and facing unit pricing

Residential consumers living in houses without unit pricing

Nonresidential consumers classified as hazardous waste generators

Nonresidential consumers classified as nonhazardous waste generator

Each of these categories faces a distribution of costs associated with each of the relevant EOL options.

In the simulation model we use Monte-Carlo techniques to predict how the population of obsolete monitors owned by consumers in each of the six categories is allocated across the different end-of-life options. The Monte Carlo simulations are conducted outside of COMPS using Microsoft Excel. The results of EOL allocations using the Monte Carlo simulation approach are then returned back to COMPS to calculate the private and social costs associated with such allocations.

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[14] A commercial or industrial establishment that disposes of 220 lbs. of hazardous material per month, equivalent to roughly seven monitors, can avoid RCRA requirements. Disposing of roughly 85 monitors per year puts an establishment into the regulated hazardous waste generator category. This threshold ignores other types of hazardous waste that might be generated.

[15] In some states, such as Massachusetts, CRTs are not allowed in landfills. In this paper, we consider the landfill ban as a policy case.

The Monte-Carlo model works as follows: we conduct separate simulations for each category of consumer and construct a representative sample of 2,000 individual members of each category using repeated draws from the relevant cost distribution for each end-of-life option,[16] with each member characterized by a single draw. After a complete set of cost parameters is selected to represent the options available to a hypothetical individual monitor owner, the model identifies the end-of-life option with the lowest cost. At that point, the simulation assigns the hypothetical owner's used monitor to the lowest cost option selected. This process is repeated 2,000 times assigning each round of draws to the lowest cost option. From the set of 2,000 draws we derive a distribution of monitors across EOL options for the sample of 2,000. The process is repeated for each of the six categories of consumers. The Monte-Carlo results are then reported back to the COMPS model, where they are used to calculate total private and external costs of monitor EOL management across the entire population of used monitors.

The COMPS model is implemented using a software package called Analytica. Analytica is a very powerful tool for conducting uncertainty analysis. Since there are huge uncertainties and variabilities associated with the parameters used in our model, Analytica is a perfect tool for our modeling purpose. The COMPS model includes four major modules. These modules are the CRT and Material Flow Module, the Cost Module, the Environmental Impact Assessment Module and Parameters, and the Index Module as shown in Figure 5. The Materials Flow Module and the Cost Module interact in the allocation of retiring CRTs among end-of-life options using the Monte Carlo approach described above, which is coded in Excel, a separate program that is used by Analytica.

[16] We assume that the cost distributions for each end-of-life option are independent of one another.

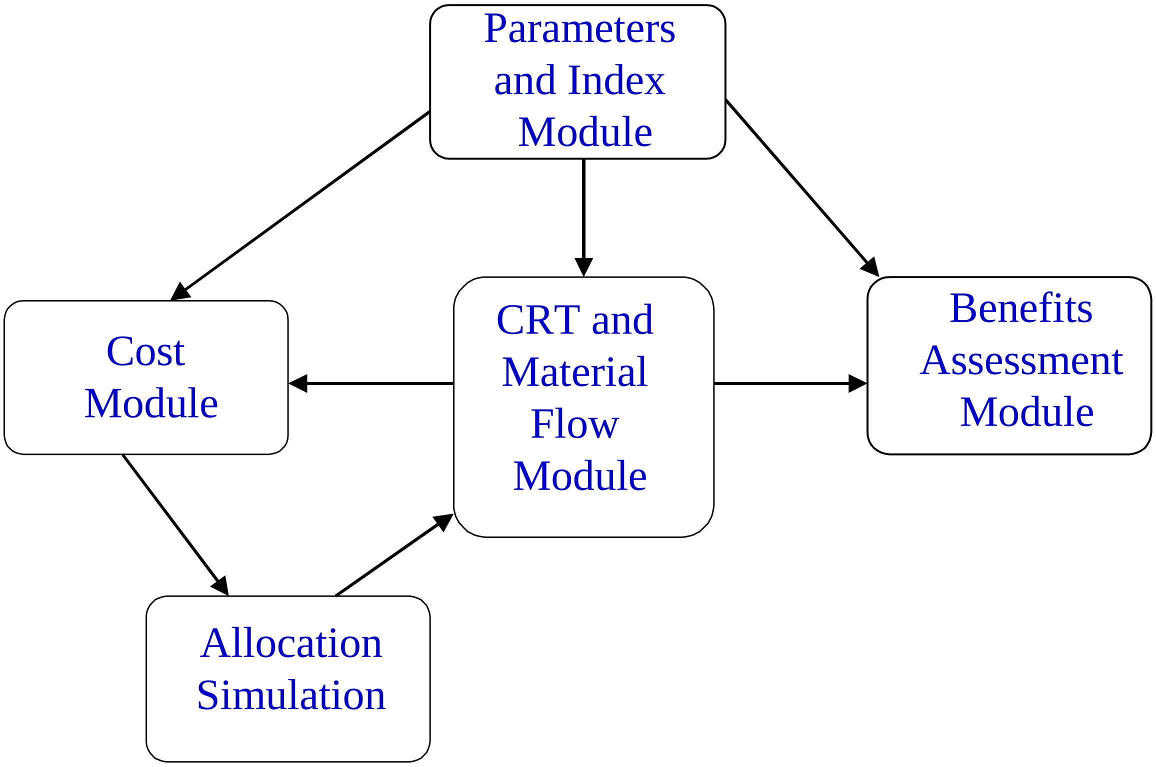
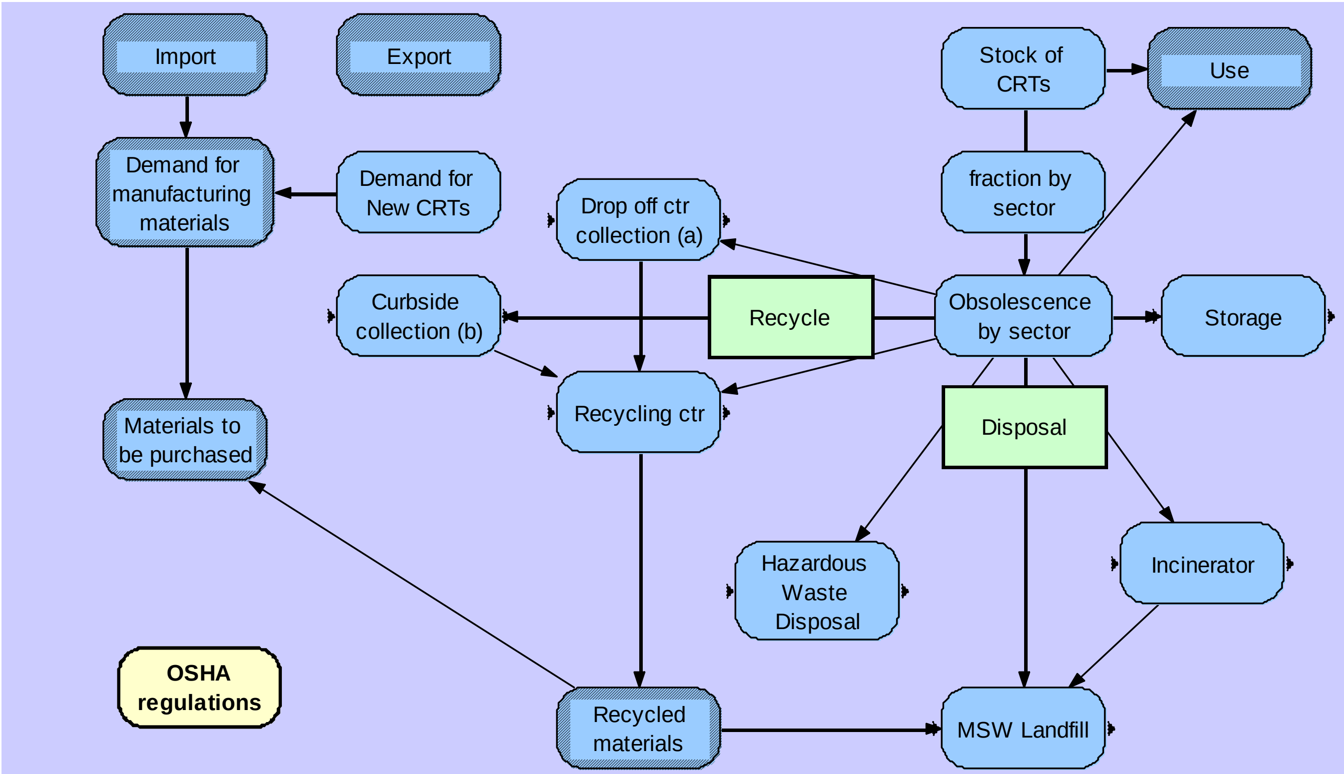


Figure 5: Overview of COMPS Model

B.1. CRT and Material Flow Module

The "full model" of the lifecycle of CRTs purchased in the United States and associated material flows—focusing on lead—is illustrated in Figure 6. Due to severe data limitations on the export of recycled materials and the demand for manufacturing materials, we illustrate these elements in the figure for completeness, but we do not include them in our approach. Thus, in this module, we focus exclusively on the discard choice. As noted above, we also do not exercise our model for the options of take-back or charitable donations.

Figure 6: Overview of Materials Flow Module of COMPS Model



(a) Not available to nonresidential sector
(b) "Turned off" as a policy available to residential and nonresidential sectors

B.2. Cost Module

The cost module tracks the costs associated with the various end-of-life options. Table 3 summarizes the costs for each EOL option for each consumer sector. The table identifies the types of information used to construct collection, transportation, and processing costs of the various end-of-life options. For those categories where data are sparse or nonexistent, the table describes the assumptions we made to develop estimates of these costs based on cost estimates for other options.

We include both the private and social costs for each end-of-life option by sector. The private costs include residential household time and travel (transportation) costs; shipping costs if the monitor is shipped to a disposal facility; and recycling process costs when these are paid by the consumer. Under some options, some of these costs are paid by general governmental revenues and we define these as the social costs for managing the waste. Social costs also include the health effects of monitor disposal (we omit health and environmental effects associated more generally with the collection and transport of waste).[17] Because of the importance of understanding the role of these health effects as a focus of our model, we discuss them more fully in the description, below, of our next module.

For all of the end-of-life options, the cost data are both highly uncertain and highly variable across the country. Table 4 describes the information we use to parameterize the distributions of costs, and a fuller discussion follows in our description of our data.

B.3. Environmental Impact Module

The environmental impacts are mainly human health damage from exposure to lead through air emissions from incinerators, and groundwater contamination at landfill sites and potential subsequent drinking water contamination; we focus on health damages associated with lead emissions from incineration of CRTs. Section V describes this part of our model in detail.

[17] Ley, Macauley, and Salant (forthcoming), for instance, discuss these externalities at length and find that they are likely capitalized into the fees charged for using landfills and incinerators as well as the "host fees" that many communities now negotiate when the facilities are sited.

Table 3: Components of cost for different CRT "end of life" options

	Residential		**Nonresidential**		
	Private cost	External cost: waste handling and health effects*	Private Cost	External cost: waste handling and health effects*	Notes
Storage	$0 if single family house; Based on mkt rent if multi-family dwelling	None	Depends on real estate cost	None	Monitor storage requires 4 square feet of space; monitors can be stacked 5 high for nonresidential storage.
MSW landfill or incineration (includes collection and transport)	$0 to 90% of households; 10% of households have UP	MCC + TC + $TF_{lf\ or\ inc}$ + $ENVC_{inc}$	PCC + TC + $TF_{lf\ or\ inc}$	$ENVC_{inc}$	Assume environmental cost of landfill = 0.
HW landfill (includes collection and transportation)	NA	NA	$PCC+TC+TF_{hwlf}$	None	This option is only relevant for 1% of nonresidential monitor CRTs.
Drop off recycling collection	HHT&TC (assume at landfill)	oper cost of drop-off site (and ads)	NA	NA	Drop-off site assumed to be co-located with landfill. Residential only.
Direct shipment to recycling center	RRSC	None	BRSC	None	Costs of packing and shipping to recyclers.
Demanufacturing and recycling process	DMFC_R + RPSC_R	None	DMFC_NR + RPSC_NR	None	External costs ignore recycling process emissions.

Key: HHT&TC= household time & travel cost	PCC= private collection cost	UP = unit pricing	lf = landfill
RRSC=Residential recyclables shipping cost	DMFC = demanufacturing cost	_R = residential	inc = incinerator
TC = Transportation cost for waste	ENVC = environmental cost	_NR = nonresidential	TF = tip fee
BRSC=Business recyclables shipping cost	RPSC = recycling process cost	NA = not applicable	HW = hazardous waste
MCC = municipal collection cost	MSW = municipal solid waste	hwlf = hazardous waste landfill	

*We ignore environmental externalities associated with collection and transport for all end-of-life options.

Table 4: Parameterization of cost distribution for different CRT "end of life" options

	Residential		Nonresidential		
	Private cost	External cost: waste handling and health effects*	Private cost	External cost: waste handling and health effects*	Notes
Storage	distribution of apartment rental values	ND	distribution of commercial real estate values	ND	
MSW landfill or incineration (includes collection and transport)	distribution of UP values for households with UP from Jenkins et al. (2000)	range of collection and transport costs from Ley, Macauley and Salant (forthcoming); distribution of health costs for incineration based on assumptions about exposure and population characteristics in health model.	range of collection and transport costs from Ley, Macauley and Salant (forthcoming); + range of tipping fees across regions	distribution of health costs for incineration based on assumptions about exposure and population characteristics in health model.	assume environmental cost of landfill option = 0
HW landfill (includes collection and transportation)	NA	NA	range of estimates for small and large quantity generators	ND	
Drop off recycling collection	distrib of distances to landfill and time costs	ND	NA	NA	Drop-off site assumed to be co-located with landfill.
Direct shipment to recycling center	range of RPSC from literature	ND	range of BRSC from literature	ND	RPSC=BRSC
Demanufacturing and recycling process	range of DMFC from literature	ND	range of DMFC from literature	ND	

Key: HHT&TC= household time & travel cost UP = unit pricing DMFC = demanufacturing cost
BRSC=Business recyclables shipping cost RPSC = recycling process cost NA = not applicable
ND = no distribution MSW = municipal solid waste HW = hazardous waste
*We ignore environmental externalities associated with collection and transport for all end-of-life options.

C. Data Sources

We have chosen to estimate our model for the stock of monitors disposed of in 1998, largely as a result of data availability. The data for the materials flow and cost modules of the COMPS model come from a variety of sources.[18] We undertook a fairly comprehensive search of prior studies on monitor disposal and recycling to learn about the material composition of monitors and the various ways they are handled at the end of their lives.

This section provides a brief sketch of the types of data sources we employ in the study. Additional details are in Appendix C.

Monitor Ownership

Because different disposal options are available to residential and nonresidential (including government) consumers, we use final computer sales data (Bureau of Economic Analysis 2000) for different sectors in the United States to determine the percentage of consumers in each category. Based on the computer sales data, we assume that residential consumers comprise 15.8 percent of all consumers in our sample. Nonresidential consumers make up the remaining 84.2 percent.

Quantity of Monitors Disposed

We base our estimate on a time series of monitor sales data (reported in National Safety Council 1999). The number of monitors disposed in a given year depends on the stock of monitors held by our consuming sectors and, from the scrappage function, the age of the monitors, the price of new monitors, disposal costs, and technological and physical depreciation. As we noted earlier, we use a combination of survey data and monitor price data to estimate the number of monitors being disposed in 1998 by these sectors.

Cost Data

Our information on the cost of different end-of-life options comes from a variety of sources. For example, most of the estimates of the costs of drop-off recycling programs come

[18] The data sources for the environmental damages module are discussed in section V.

from cost estimates generated for local pilot programs conducted primarily for data gathering purposes (EPA 1998).[19] Estimates of the costs of processing CRTs collected for recycling come from industry surveys of electronics recyclers (ICF 1997). To capture geographic and other regional differences (for instance, landfill and incineration fees vary significantly by locality, as do many of our other cost variables), we were able to compile these data for three regions that are representative of the range of costs in the United States: the Northeast (Massachusetts, Maine, Rhode Island, Connecticut and New York); Washington, D.C., metropolitan area (Washington, D.C., and the surrounding areas in Maryland and Virginia); and the Midwest (Nebraska, Missouri, Iowa, Illinois, Kansas, and Arkansas).

Residential Storage Costs

We assume that residential storage costs differ for those individuals living in a house and those living in an apartment.[20] We assume that storage costs are zero for residents living in a detached house due to the ample storage space generally available in those units. We assign apartment dwellers a positive storage cost based on rental rates per square foot, under the assumption that these residents have limited storage space available.

Nonresidential Storage Costs

We calculate a distribution of nonresidential storage costs for selected cities in our three regions using rental rates per square foot and estimates of the square feet used for monitor storage.

Residential Waste Handling Costs

The costs of waste handling and disposal vary among residential consumers depending upon the type of pricing used in their community. The vast majority of individuals in the United States face essentially zero marginal costs for waste handling. These individuals do not have unit pricing and generally pay a flat rate per month or year for garbage removal. Those consumers who have to pay a unit price per container or bag for garbage removal will experience waste-handling costs based on the volume of garbage disposed. We assume that the waste is

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[19] In the model we assume there is no fee associated with drop-off recycling.

[20] Based on US Census Bureau data for 1990, we assume that 59% of the national population lives in a detached home while the remaining 41% live in an attached home or multi-unit dwelling.

hauled either to landfill or incineration facilities based on decisions made by the waste hauler. We use the national average percentage of waste sent to landfills and to incinerators to represent this allocation.

Nonresidential Waste Handling Costs

The waste handling costs of nonresidential consumers vary depending upon the amount of waste generated. Businesses and government agencies that generate a large quantity of CRT waste, or of total hazardous waste including CRTs, are subject to hazardous waste disposal requirements under the Resource and Conservation Recovery Act (RCRA) subtitle C, while those with smaller quantities of hazardous waste, including CRTs, are exempt. For CRTs, the RCRA requirement limits storage of used CRTs, enforces record-keeping requirements for shipments of used CRTs, and stipulates that CRTs must be disposed in a hazardous waste disposal facility or treated to make them nonhazardous before disposal in a regular landfill. These consumers are not legally permitted to dispose of their CRTs as municipal solid waste (thus incineration and disposal in a MSW landfill are not available as options in model). Small quantity generators, which produce between 100 and 1,000 kg of hazardous waste per month, are subject to a less restrictive set of hazardous waste disposal requirements. We assume that 1% of nonresidential consumers are subject to RCRA subtitle C, and the remaining 99% are not.

Nonresidential consumers that are small quantity generators of hazardous waste have the option of disposing of their CRT waste by incineration or landfilling. Total disposal costs include the collection and transportation costs to the landfill or incinerator and the fee paid at the landfill or incinerator.

Residential and Nonresidential Recycling Costs

Residential consumers in apartments and homes have the option of recycling their monitor either by taking it to a drop-off center to be sent to a recycling center at a later date or by sending it directly to a recycling center. [21] The costs involved in taking the monitor to a drop-off center include the traveling costs incurred and the opportunity costs of traveling time. We assume that nonresidential consumers face the same costs as residential consumers to send monitors directly to a recycling center.

[21] Residential consumers in many localities also may leave their CRT at the curb for pick-up and subsequent recycling; we are in the process of modeling this option

V. Modeling the Environmental Damages

Lead, cadmium, and mercury are the main toxic elements found in CRTs (Biddle 2000). Lead is generally the main focus of concern due to the large amount contained in the glass face of the CRT. Disposal of CRTs through incineration or landfilling can result in the release of lead into the environment. CRT glass may break when a computer monitor or TV is placed in a landfill, potentially resulting in lead contamination of groundwater from landfill leachate.[22] Incinerated lead may be an even bigger problem, as lead contained in the glass either is emitted into the air or remains in the ash. The ash obtained from the incinerator must then be disposed of in a landfill—or a hazardous waste landfill if the lead content is above acceptable levels.

A. Lead Emissions

Lead uptake may result in several health problems for different segments of the population. Health effects experienced by both adult men and women may include hypertension, stroke, possible cancer, and premature death. Men also may experience nonfatal coronary heart disease, and women may experience heart attacks and reproductive effects. Infants and children experience different effects. Problems may include decreased IQ and gestational age, reduced birth weight, other neurological and metabolic effects, and possible increases in infant mortality. Children may experience behavioral changes, metabolic effects, interference with growth and nervous system development, anemia, and possible cancer (RCG/Hagler Bailly 1994; Wade Miller Associates Inc. and Abt Associates 1991; and EPA 1986).

As we pointed out in the previous section, while lead exposure from both air emissions and groundwater pollution can result in these adverse health effects, the environmental damages segment of our model focuses on the health impacts from lead air emissions. We exclude lead exposures associated with emissions into groundwater from landfill leachate and associated with subsequent drinking water contamination because these emissions are likely to be small and their effects on drinking water are likely even smaller. Information from two sources (Townsend 1999; McKenna 1996) suggests that the emission rates of lead from CRTs in landfill leachate could range between 4.1 E-10 lbs/CRT and 3.5 E-3 lbs/CRT. The top end of this range

[22] While most landfills in the United States have a leachate collection system, the contamination problems will still not be completely alleviated. Also, other countries that do not have leachate collection requirements or systems may face greater risks of this type of contamination.

represents the result of grinding CRT glass into very small pieces, thereby mobilizing more of the lead than would typically by released, and assumes that all of the leachate is released into the environment immediately and not contained. Concentrations toward the middle or lower end of the range are more likely under typical disposal conditions, where glass pieces tend to be larger. In addition, much of the lead is likely to be absorbed by the soil during groundwater transportation, thereby attenuating the effect on drinking water in wells. Exposure to lead in contaminated wells can also be reduced through testing and taking preventative actions such as using a water filter or switching to bottled water (for example, see discussion in Skipton and Hay, 1997). In contrast to the water contamination situation, air emission rates from CRT incineration tend to be much higher (.00026 lbs/CRT) and averting actions to limit exposure are not available.

The specific health effects are described in the data section. The remainder of this section describes the environmental impact section design (see Figure 7 for the environmental impacts model diagram).

We use the total number of incinerated computer monitors estimated in the Material Flow portion of the model to calculate the health damages due to emissions. The Material Flow section allocates the total number of discarded monitors to different end-of-life options. The number of monitors disposed of by incineration is used to calculate the average total lead emissions due to CRT incineration. Average emissions are then used to calculate the average change in ambient concentration of lead in the air.

To translate the increase in air emissions of lead into health effects, the change in ambient emissions must be converted to the average increase in blood lead among members of the different exposed populations. The average increase in blood lead is used in the estimation of dose response functions and number of incremental cases of certain health conditions. The dose response functions estimate the change in probability of the health condition using the baseline blood lead and the post-exposure blood lead. The incremental number of cases per year is generally estimated using the change in probability of the health condition for any exposed individual, the size of the affected population and percent of the distribution exceeding a certain blood lead level. Damages for each of the affected groups are estimated by multiplying the incremental number of cases by the value per event. The damage valuation for each health condition is estimated and presented in the summary of annual damages.

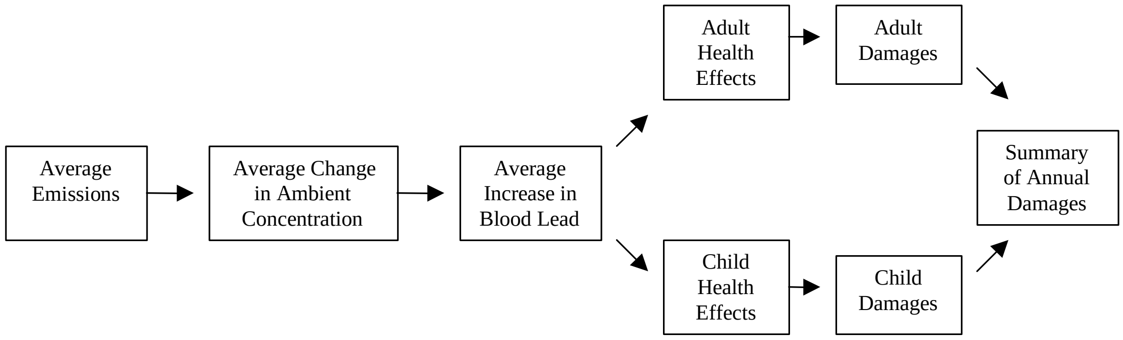

Figure 7: Diagram of the Lead Health Effects Submodel

B. Data Sources

The majority of data used in the lead health effects portion of the model is from the RCG/Hagler Bailly New York State Environmental Externalities Cost Study (1994). Data from the U.S. Environmental Protection Agency (EPA), U.S. Census Bureau, and data estimated in other parts of our model are also used. The main input to the RCG/Hagler Bailly health estimates is a lead impact and damage model used by the EPA. The model originally was created to estimate the impacts of lead in gasoline. It was further developed by some consulting firms and then used for other lead impact analyses (RCG/Hagler Bailly 1994).

Average emissions data were derived using the national lead emissions from CRT incineration in the Materials Flow portion of our model. Total national emissions were divided by the total number of waste-to-energy facilities and incinerators in the United States (Franklin Associates 1998) in order to estimate the average lead emissions from CRT incineration. The average change in ambient air lead concentration was derived using the average emissions estimation and the average change in ambient concentration of air lead due to a municipal solid waste incinerator came from the RCG/Hagler Bailly study. In order to apply the changes in ambient air lead concentrations to health effects due to lead, we used the changes in air lead levels to estimate the average increase in blood lead. Changes in blood lead are calculated using the relationship between average change in ambient air concentration and average increase in blood lead specified in the RCG/Hagler Bailly study.

The estimates of average increase in blood lead are then applied to a host of different health effect estimation functions for different health conditions. Dose-response functions relating changes in blood lead to specific health conditions were available only for certain health conditions and certain populations. The RCG/Hagler Bailly study did not include all possible health effects or the effects on women or older youth due to lack of data or inconclusive evidence. Only those health effects used in the RCG/Hagler Bailly damage valuations are included in our model. Health effects experienced by adult males included in the model are hypertension, nonfatal coronary heart disease, stroke (including cerebrovascular accidents and initial atherothrombotic brain infarctions), and premature death. Health impacts on children and infants contained in the model include decreases in IQ, increased probability of mental retardation, cognitive damage associated with blood levels of greater than 20 micrograms/dl, and asymptomatic infants and children in five different risk groups.

Dose response functions and the number of incremental cases of each condition per year are used in the estimation of health damages due to lead exposure. The dose response functions use the average increase in blood lead and the baseline blood lead to determine the change in probability of a health event occurrence (for adult health effects). The incremental number of cases for each health condition or event depends on the size of the exposed population. The population estimates used are from one of our three sample areas: the Washington, D.C., metropolitan area, the midwest region, and the northeast region. The percentage of the population in the affected group is multiplied by the sample population to obtain the affected population for each health condition. All population data were obtained from the 1990 U.S. Census of Population (U.S. Census Bureau 1990).

The incremental number of cases per year is used in the estimation of the annual damages for each health event. Dollar estimates of value per event were obtained from the RCG/Hagler Bailly study. For several of the health conditions, including strokes and coronary heart disease, these values are based on estimates of medical costs of treating the condition which have been adjusted by a factor greater than one to capture missing components of cost. For other health conditions, such as loss of IQ points, the value is based on lost future earnings and additional costs of educating mentally retarded youngsters. For each health condition, the relevant values are multiplied by the incremental number of cases per year to obtain total annual damage values.

VI. Limitations of the COMPS Model

The RFF COMPS model is the only model we are aware of that can simulate the CRT lifecycle and end-of-life management decisions and track the environmental damages associated with potential lead releases from CRT disposal. Despite its uniqueness and usefulness for policy analysis, the model has several limitations. Most of these limitations are due to extremely limited data. The U.S. government collects data on purchases, imports, and exports of electronic products, but CRTs are not identified as a separate category. Thus, we rely on private survey estimates of the size of the CRT market, the stock of existing CRTs and the number of CRTs that become obsolete in a given year, together with whether these data make sense in the context of the scrappage function. Information on what happens to CRTs once they reach the end of their useful life also is extremely spotty, and estimates regarding how many CRTs are stored, recycled, donated, or disposed of vary considerably across sources. Fortunately for us, the Monte Carlo approach we use allows us to make use of those variations in our simulation analysis.

Our analysis also is limited because we don't consider CRTs found in televisions or the role of flat panel displays as substitutes for traditional computer monitors. In the United States, the television market is comparable in size to the market for computer monitors (National Safety Council 1999). However, televisions generally have a longer life span than do computer monitors. Also, we believe that decisions about what to do with computer monitors at the end of their lives are fundamentally different from decisions about how to handle televisions, and thus an analysis of television lifecycles would require a different model. We exclude flat panel displays (also known as liquid crystal displays or LCDs), which could be a substitute for regular monitors with CRTs, because these displays currently cost up to 10 times as much as a regular CRT monitor of similar size. Thus, a policy aimed at limiting CRT disposal would have to impose substantial cost on a CRT user before it would lead her to switch to a flat panel. Ultimately, we would like to incorporate information on flat panels into the model so we could simulate the effect of policies aimed at CRTs on the flat panel market under different assumptions about the prices of flat panels. Eventually, growth in the market for flat panels also could have an adverse effect on demand for recycled CRT glass.

The COMPS model also is limited in its ability to address DfE issues. One of the goals of product take-back requirements for manufacturers is to provide them with an incentive to redesign their products so they will be easier to recycle at EOL. The current version of the model has a representation of computer manufacturers and it does not categorize CRTs by ease of recyclability. Ultimately, we would like to enhance the model in this regard and, to that end, we offer discussion of these approaches in Appendix E.

Finally, because our analysis is focused on CRTs, we don't track what happens to the other components of computer monitors, such as the plastic housing or cover. For example, incineration of the plastic housing can release dioxins according to some researchers (although this conclusion is controversial), and we do not include this effect in our environmental damages module. We also don't address the feasibility or costs of plastics recycling. If recycling of plastics from computers can be done cost-effectively, that is plastics can be recycled more cheaply than they can be disposed, then our model may mischaracterize the economics of CRT recycling. In this case, failure to incorporate plastics into the model could mean that our estimates of the costs of CRT recycling exceed the costs of collecting and recycling entire computer monitors. However, whether or not cost-effective recycling of plastic from used computer monitors is feasible is still unclear (American Plastics Council 2000).

VII. Results

In this section, we discuss the model results for a baseline case and several policy scenarios.

A. The Parameters of the Basic Model

Several parameters and assumptions are basic to the COMPS model and each of the policy scenarios we address. These elements are the consumer categories and the allocation of the stock of end-of-life monitors among these categories of consumers.

As described in section IV d, we derived the private and social disposal cost distributions for all of our consumer groups and disposal options. To summarize these distributions, Tables 5 and 6 list the means and standard deviations for private and social costs per monitor.

Table 5: Baseline Case Mean Private Costs (1998 $)

Consumer EOL option	Residential Consumers				Nonresidential Consumers	
	Apartment dwellers		House dwellers		Hazardous waste generators	Nonhazardous waste generators
	with unit pricing	without unit pricing	With unit pricing	without unit pricing		
Storage	26.80 (10.55)	26.80 (10.55)	0	0	18.65 (4.14)	18.65 (4.14)
Incineration	3.00 (1.08)	0	3.00 1.08	0	NA	1.03 (0.09)
Landfill	3.00 (1.08)	0	3.00 (1.08)	0	NA	0.72 (0.27)
Drop off	22.53 (9.04)	22.53 (9.0 4)	22.53 (9.04)	22.53 (9.04)	NA	NA
Hazardous	NA	NA	NA	NA	6.06 (0.38)	NA
Recycling	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)

Notes: Standard deviations are listed in parentheses. NA = Option is not available.

Table 6: Baseline Case Mean Social Costs* (1998 $)

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Nonhazardous waste generators
Storage	26.80 (10.5)	26.80 (10.55)	0	0	18.65 (4.14)	18.65 (4.14)
Incineration	3.00 (1.08)	1.03 (0.09)	3.0 (1.08)	1.03 (0.09)	NA	1.03 (0.09)
Landfill	3.00 (1.08)	0.72 (0.27)	3.00 (1.08)	0.72 (0.27)	NA	0.72 (0.27)
Drop off	22.53 (9.04)	22.53 (9.04)	22.53 (9.04)	22.53 (9.04)	NA	NA
Hazardous	NA	NA	NA	NA	6.06 (0.38)	NA
Recycling	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)	25.50 (7.55)

Notes: Standard deviations are listed in parentheses. NA = Option is not available.
*Social cost is the sum of private cost and external cost, but does not include health effects, which are non-linear in quantity of CRTs incinerated.

B. Baseline Case

We first use Monte Carlo simulation to estimate monitor discard choices for each group of consumers, assuming that each chooses the least costly discard option available to them. Tables 7 and 8 display, respectively, the estimated percentage and actual quantity of monitors discarded for each option by type of consumer. In the baseline, we use the nationwide percentage of municipal solid waste sent to landfills and incinerators, to allocate monitor disposal by incineration and landfill (the national averages are 22% to incinerators and the remaining 78% to landfills).

The results reported in these tables show that storage costs are an important determinant of what happens to used monitors. For apartment dwellers, storage is almost always more expensive than disposal, even when an apartment dweller faces a unit price for waste management and collection. However, for house dwellers, storage is assumed to be free and therefore house dwellers will almost always select storage if there is a fee for disposal. However, when disposal is free, notably fewer monitors are stored (only one third) and the majority of the rest are thrown away. Nonresidential consumers also almost always find that disposal is cheaper than storage. Fewer than 0.5% of the monitor consumers who are classified as hazardous waste generators find it economical to store their used monitors and none of the nonhazardous waste generators select storage. Recycling also is rarely selected in the baseline case.[23]

[23] Our results concerning recycling are at odds with the data reported by the National Safety Council (1999) for 1998. In their report they find that roughly 10% of all obsolete CRTs were recycled in 1998 with virtually all of the recycling activity occurring in the nonresidential sector. Given the cost parameters in our model, our results suggest that such a level of recycling activity is not economically justified in the absence of recycling subsidies or a ban on CRT disposal. In policy scenario C, reported below, we look at what level of recycling subsidy might be necessary to generate a 10 % recycling rate in our model.

Table 7: Baseline Case EOL Allocation by Consumer Group (%)

Consumer EOL option	Residential Consumers				Nonresidential Consumers	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	1.05	0.85	98.50	33.02	0.35	0
Incineration	21.33	21.60	0.20	14.53	NA	12.70
Landfill	75.62	76.60	0.35	51.51	NA	87.25
Drop off	1.55	0.90	0.90	0.90	NA	NA
Hazardous	NA	NA	NA	NA	99.20	\NA
Recycling	0.45	0.05	0.05	0.05	0.45	0.05

Note: NA = Option is not available.

Table 8: Baseline Case EOL Quantity Allocation (Number of CRTs)

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	1,078	7,924	144,700	438,200	466	0	592,368
Incineration	21,910	201,400	294	19,2800	NA	1,674,000	2,090,404
Landfill	77,660	714,100	514	683,600	NA	11,500,000	12,975,874
Drop off	1,592	8,390	1322	11,940	NA	NA	23,244
Hazardous	NA	NA	NA	NA	132,000	NA	132,000
Recycling	462	466	73	664	599	6,589	8,853
Total	102,702	932,280	146,904	1,327,204	133,064	13,180,589	15,822,743

Note: NA = Option is not available.

Policy Scenario A: Ban Incineration and Landfill Disposal

Some jurisdictions, such as the state of Massachusetts, have enacted disposal bans to address the potential health effects associated with CRT incineration and the lead contamination from CRTs in municipal solid waste landfills. In policy scenario A, we impose such a ban in our model to see what effect it has on the distribution of end-of-life monitors. In this analysis, the discard option of hazardous waste processing is only available to nonresidential consumers of larger quantities of used monitors.

We find that a disposal ban increases the aggregate monitor-recycling rate to about 23%, with most of the recycling done by the nonhazardous waste generators in the nonresidential sector and by apartment dwellers. House dwellers that face zero storage costs continue to prefer storage to recycling. Nonetheless, the aggregate recycling rate among all residential consumers is about 30% with this policy in place.[24] Most hazardous waste generators continue to dispose of their used monitors since, as shown in table 1, the cost of doing so is almost always less than the cost of recycling.

Table 9 reports the costs and rates of recycling associated with the different policies. Primarily as a result of the high costs of storage, the incremental private costs of a disposal ban are roughly $280 million. This incremental cost is nearly two orders of magnitude greater than the value of the avoided health effects arising from eliminating all CRT incineration. However, there may be other benefits, both at the disposal stage and upstream in the CRT lifecycle that are

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[24]This estimate is roughly consistent with the recycling behavior that we observe under the CRT landfill ban in Massachusetts. When the state of Massachusetts banned CRT disposal, it also implemented a number of CRT drop-off programs to collect CRTs for recycling. During the first seven months of the ban (April to October 2000), roughly 40,000 CRTs, including both televisions and computer monitors, were collected for recycling (Personal communication, MA Department of Environmental Protection). Using census information on the total number of households in Massachusetts in 1998 (2,349,000) and on the percentage of households with a computer in 1997 from the Census (36.6%), we estimated that 859,734 Massachusetts households have computers and we assume that same number have monitors. Based on National Safety Council data on CRT shipments and retirements over time, we calculate that 10% of the cumulative number of the CRTs shipped between 1992 and 1998 (the number in the household in 1998) became obsolete in that year. If the 40,000 CRTs recycled by households in the first seven months of the ban were all from computer monitors and were all from the pool of CRTs that became obsolete in that year (and not CRTs being pulled out of storage), then approximately 46.5 % of the roughly 86,000 CRTs owned by residential consumers that became obsolete in Massachusetts in 1998 were recycled. The fact that the CRTs collected for recycling probably include a mix of televisions and computer monitors as well as a mix of recently obsolete CRTs and CRTs taken from storage, helps to explain the difference between the two estimates.

not accounted for in this analysis. Nonetheless, this finding suggests that these benefits would have to be quite large to justify the costs of the disposal ban.

Table 9: Policy Costs and Recycling Rate Results

Scenario	Total Private Cost (millions $)	External Cost: Waste Handling (millions $)	External Cost: Health Effects (millions $)	Cost of Subsidy (millions $)	Recycling Rate (%)
Base Case	12.14	1.4	2.67	NA	0.2
A	292.3	0.0	0	NA	23.4
B	333.7	0.0	0	96.6	61.1
C	50.8	1.1	2.36	22.9	10.3
D	100.0	0.8	2.03	65.9	23.1
E	49.3	1.0	0	NA	3.5
F	11.8	16.3	2.5	NA	5.3
G	267.4	33.2	0	NA	29.7

Note: NA = Subsidy was not involved in the policy scenario.

A: Ban Incineration and Landfill Disposal
B: Ban Disposal and Subsidize Recycling by $10 per CRT
C: Subsidize Recycling to Achieve 10% Recycling Rate
D: Subsidize Recycling to Achieve 23% Recycling Rate
E: Ban Incineration Only
F: Curbside Recycling
G: Curbside Recycling with Ban on Incineration and Landfill Disposal

Policy Scenario B: Ban Disposal and Subsidize Recycling by $10

In this scenario, we ban the incineration and landfill disposal options for all generators and we reduce the mean of recycling cost (either by means of direct shipment to recycling centers or by the use of drop-off recycling programs) by $10 per monitor. This is intended to represent the effects of a subsidy to monitor recycling, in combination with a landfill ban. The purpose of this scenario is to see how high the recycling rate will be in this case and what the costs of such a policy would be.

We find that this policy increases the monitor-recycling rate to just over 60%. This policy increases recycling by all categories of consumers, but, like policy scenario A, it has a particularly strong effect on apartment dwellers and on nonresidential consumers that are not classified as hazardous waste generators. Within the former category of consumers, the recycling rate rises to over 90%, while in the later group it rises to about 65%. (See Appendix D Table D-B1.)

The costs of this policy are substantial. Annual private costs are in excess of $300 million and the cost of the subsidy itself is slightly less than $100 million.[25] Comparing this scenario to scenario A suggests that the incremental cost of using the subsidy to raise the recycling rate from 23% to 61%, in the presence of a disposal ban is roughly $200 million.

Policy Scenario C: Subsidize Recycling to Achieve 10% Recycling Rate

In policy scenario C, we use the COMPS model to show that a recycling subsidy of $14 per monitor will achieve a 10% aggregate recycling rate, including both those taken to drop-off centers and those shipped to commercial recyclers. In the absence of a disposal ban, many of the monitors that are recycled would have been stored or disposed of as hazardous waste. As a result the health benefits associated with this policy are only $300,000, roughly 1/8 of the $2.67 million in health benefits arising from a landfill ban.

Policy Scenario D: Subsidize Recycling to Achieve 23% Recycling Rate

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[25] If this subsidy is paid for out of tax revenues, the cost to the economy will be even higher than the cost noted here, since there will be efficiency losses on the order of 25% associated with using public funds to pay the subsidy (estimates of the loss vary among researchers in public finance, but most agree that the costs are substantial. For example, see discussion in Ballard and Fullerton, 1992, and Feldstein, 1999).

From scenario A, banning disposal of monitors leads to an aggregate recycling rate of 23%, but at an incremental cost (above the baseline case) of nearly $280 million. We now undertake a scenario in which we identify the level of a recycling subsidy policy that would achieve the same recycling rate without an incineration and landfill ban. We find that an average subsidy of $18.00 per CRT will produce a 23% recycling rate. The incremental cost of this policy (including the cost of the subsidy) is about $156 million, slightly more than one half of the cost of the disposal ban. However, the health benefits associated with this policy are less than 25% of the health benefits associated with the ban.

Policy Scenario E: Ban Incineration Only

Here we model a ban on monitor incineration only. Because our analysis of the literature on health damages associated with lead disposal suggests that the benefits from reduced disposal (ignoring benefits from reduced disposal of other components or at other points upstream in the CRT lifecycle) arise largely from reduced incineration, we consider a limited disposal ban that targets incineration only. When we model this ban, we assume that consumers cannot divert their monitors from an incinerator to a landfill, but must instead find other discard options.

We find that this policy option has the same environmental benefits at the disposal stage as policies A and B, but at a much reduced cost. Indeed, the incremental cost of policy E relative to the baseline is about $38 million, roughly an order of magnitude lower than the incremental costs of policies A and B. The aggregate recycling rate associated with this policy is only 3.5%. However, given that recycling is the most costly discard option, this low recycling rate contributes to the low cost of this policy.

Policy Scenario F: Curbside Recycling

In this scenario, we model a free curbside recycling program only for households and similar to the program in place in Minneapolis, MN, for all types of consumer electronics. This program substantially lowers the time and out-of-pocket costs to households of recycling their CRTs by providing for free collection at the home.

We find that this policy yields a little more than a 5% aggregate recycling rate for CRTs. All of the additional recycling relative to the baseline under this policy is coming from households and the policy leads to significant increases in recycling among residential CRT users, particularly those who live in apartments or who pay unit prices for trash disposal. However, because households account for only 15% of obsolete CRTs, the effect of changes in recycling by households has a small impact on the overall recycling rate.[26] Also, the increase in recycling is coming largely out of storage and thus there is virtually no change in the environmental damages associated with incineration of CRTs under this policy. The total cost of this policy (including the external cost) is roughly $30 million, $14 million above the baseline.

Policy Scenario G: Curbside Recycling Plus Incineration and Landfill Ban

This policy combines policies A and F from above. Because incineration is banned, the external costs of incineration are eliminated. Under this combined policy, the aggregate recycling rate rises to almost 30%, slightly higher than the sum of the recycling rates achieved under the two separate policies. However, with an annual price tag of about $300 million, roughly 10 times the cost of the curbside recycling program by itself, this policy is quite costly. Thus, this combined policy purchases a six-fold increase in the recycling rate and the elimination of the health costs from incineration for about $270 million per year.

VIII. Conclusions

Given the high level of public attention in the United States and in countries around the world to waste management of end-of-life CRTs, our modeling framework of consumers' discard decisions offers a basis for evaluating policy alternatives for the case of a growing component of the CRT waste stream, EOL computer monitors. The largest challenge is perhaps the lack of data and poor quality data to implement the model. For this reason, our approach characterizes much of our cost data by specifying probability distributions around values we glean from a largely anecdotal literature and based on numerous assumptions. We also lack good data on the demand for waste management emanating from different types of consumers; both their behavior and their choice of discard options for EOL monitors vary significantly. For instance, we have no data on the numbers of nonresidential consumers who generate substantial quantities of used CRTs and whose choice of discard options, at least in the United States, is severely constrained by law. Because of data limitations, we also are unable to address exports of used CRTs and the effects of CRT recycling on markets for secondary and primary lead and the cost of producing new CRTs.

[26] We assume here that business CRT owners are not going to take their used CRTs home with them for recycling at the curb, something that might happen if residential users have access to free curbside recycling and business users do not.

Despite these data limitations, our results are suggestive of the merits and disadvantages of policy approaches that are now being taken or are under consideration. The significant differences that result among the costs of our policy scenarios indicate that identifying the most cost-effective policy depends on the goal of the policy (for example, banning incineration, encouraging recycling generally, or encouraging recycling to meet a specified recycling goal). In the absence of any policy restrictions on the options for discarding a used monitor, very few monitors will be recycled although the average disposal cost, including our estimate of the value of health effects associated with incineration—is quite low (about $1 per monitor based on our data). Policies to reduce the disposal of monitors in landfills and incinerators are costly due to the high costs of storage for some consumers and of recycling as an alternative.

We also find that a recycling subsidy is a more cost-effective way to increase monitor recycling than a disposal ban. However, a recycling subsidy reduces storage more than it reduces incineration or landfilling, causing the environmental benefits from reduced incineration to be small.

While this study does not consider manufacturer take-back policies explicitly, the results do provide some lessons for the design of such policies. Specifically, the results suggest that a manufacturer take back program would require some sort of penalty for disposing of monitors with regular household waste in places where marginal disposal costs are zero in order to encourage waste diversion. Take back policies would also likely require some sort of incentive to get monitors out of storage in order to increase recycling among customers who have access to inexpensive storage.

Finally, the benefits of reducing airborne emissions of lead associated with CRT incineration appear to be small. Other end-of-life benefits or environmental benefits that may be achievable earlier in the CRT lifecycle would need to be large to justify the costs associated with policy actions that induce increased storage and recycling. If the goal of a policy is to reduce the potential health damages associated with incineration of lead, banning incineration of CRTs is much more cost-effective than banning all forms of disposal. A ban on landfilling and incineration of used monitors results in an average disposal cost of almost $20 per monitor while a ban on incineration results in an average cost of just more than $3 per monitor.

These results suggest several directions for future research. With some modifications, the model could be adapted to evaluate other policy options, such as a deposit-refund. Representing a manufacturer take back program in the model is more problematic due to the high degree of uncertainty about how such a program would function and the complete lack of data about its cost. Moreover, an important anticipated benefit of a take-back programs over other forms of managing used monitors is the extent to which such a program creates stronger incentives for making new monitors that are easier to recycle, use fewer hazardous materials, and are less waste intensive. Enhancing the model so that it can evaluate the policies along this “design for environment” dimension is an important area for future research.

References

Alberini, Anna, Winston Harrington, and Virginia McConnell. 1998. "Fleet Turnover and Old Car Scrap Policies." Washington, DC: Resources for the Future Discussion Paper 98-23.

American Plastics Council. 2000. *Plastics from Residential Electronics Recycling: Report 2000*, a report prepared by Headley Pratt Consulting for the American Plastics Council.

Ballard, Charles L. and Don Fullerton. 1992. Distortionary Taxes and the Provision of Public Goods. *Journal of Economic Perspectives*. 6(3): 117-131.

Biddle, David. 2000. *End-of-Life Computer and Electronics Recovery Policy Options for the Mid-Atlantic States*. Center for Solid Waste Research.

Campbell, Ted. 1999. South Carolina Electronic Equipment Recycling Program, South Carolina Department of Commerce. Presentation to the EPR2 Conference, March 23–24.

Caudill, R.J., V.M. Thomas, B. Kirchhoff, J. Kliokis, and J. Linton. *Lifecycle Analysis of CRTs*, New Jersey Institute of Technology. http://www.njit.edu/merc/research/reports/lca_CRT.html (Accessed December 2000).

CB Richard Ellis National Real Estate Index Market Monitor, first quarter 1998.

Clarke, Raoul. 1999. Florida's Strategy for End-of-Life CRTs and Other Electronic Equipment. Florida Department of Environmental Protection, presented to DEER2 Conference on October 26.

Clean Japan Center. 2000. *Recycle-Oriented Society: Towards Sustainable Development*. (January).

Clemmer, Steven L. Alan Nogee and Michael C. Brower. 1999. *A Powerful Opportunity: Making Renewable Electricity the Standard*. Cambridge, MA: Union of Concerned Scientists (January).

Dickson, Peter R. and Neil Peters-Michaud. 1999. *Analysis of the Cathode Ray Tube End-of-Life Management Industry in the United States*. School of Business, University of Wisconsin-Madison.

European Commission. 2000. *Explanatory Memorandum to the Directive on Waste Electrical and Electronic Equipment and on the Restriction of the Use of Certain Hazardous*

Substances in Electrical and Electronic Equipment, European Parliament and Council on Waste Electrical and Electronic Equipment (June 13).

Eustace, H, G. Goulet, D. Harden, and R. Patelli. 1995. *A Business Assessment of Electronics Recycling for The Gordon Institute*, a masters thesis, August 1.

Feldstein, Martin. 1999. "Tax Avoidance and the Deadweight Loss of the Income Tax," *Review of Economics and Statistics.*

Franklin Associates. 1999. *Characterization of Municipal Solid Waste in the United States: 1998 Update*. Prepared for U.S. Environmental Protection Agency, Municipal and Industrial Solid Waste Division, Office of Solid Waste, Washington, DC.

Goldstein, Nora. 2000. The State of Garbage in America. *BioCycle*, April 2000.

Hamilton, Bruce and Molly K. Macauley. 1999. Heredity or Environment: Why is Automobile Longevity Increasing? *Journal of Industrial Economics*, June, vol. 47.

Hileman, Bette. 2000. EU Wants Electronic and Electrical Products Recycled. *Chemical & Engineering News*, 78(28): 38—40 (July 10)

ICF Incorporated. 1997. *Environmental and Economic Impact Assessment of Cathode Ray Tube Recycling*, prepared for the Office of Solid Waste, U.S. Environmental Protection Agency (May 27).

Jenkins, Robin R., Salvador A. Martinez, Karen Palmer, and Michael J. Podolosky. 2000. The Determinants of Household Recycling: A Material Specific Analysis of Recycling Program Features and Unit Pricing. Washington, DC: Resources for the Future Discussion Paper 99-41-REV.

Lambert, Paul. 1994. *The Recycling of Cathode Ray Tubes*, Manchester Metropolitan University, available at http://sun1.mpce.stu.mmu.ac.uk/pages/projects/dfe/pubs/dfe16/report16.htm (July 15, 2000).

Ley, Eduardo, Molly K. Macauley, and Stephen Salant. Forthcoming. Spatially and Intertemporally Efficient Waste Management: The Costs of Interstate Trade Restrictions. *Journal of Environmental Economics and Management*.

Lightly, Jeanne. 2000. What to do with that old computer**.** *Des Moines Business Record*,16(5): 28 (January 31).

Massachusetts Department of Environmental Protection. Personal communication. October 31, 2000.

Matthews, Emily, Christof Amann, Stefan Bringezu, Marina Fischer-Kowalski, Walter Hüttler, René Kleijn, Yuichi Moriguchi, Christian Ottke, Eric Rodenburg, Don Rogich, Heinz Schandl, Helmut Schütz, Ester Van Der Voet, and Helga Weisz. 2000. *The Weight of Nations: Material Flows from Industrial Economies*. Washington, DC: World Resources Institute.

McKenna, D., K. Powers, S. Reagan, et al. 1996. *Life Cycle Assessment of the Disposal of Household Electronics.* Tufts University (August).

Ministry of International Trade and Industry. 1998. "Legislative System for Promoting the Creation of a Recycle-Oriented Society." (June).

Minnesota Office of Environmental Assistance (MOEA). 1995. *Management of Waste Electronic Appliances*, (July).

Mizuki, C. and G. Schuldt. *Computer Display Industry and Technology Profile*. U.S. Environmental Protection Agency. http://www.epa.gov/opptintr/dfe/compdisp/tech_reports/cdprofile.pdf (Accessed December 2000).

National Safety Council. 1999. *Electronic Product Recovery and Recycling Baseline Report: Recycling of Selected Electronic Products in the United States* (March).

Palmer, Karen and Margaret Walls. 1997. Optimal Policies for Solid Waste Disposal: Taxes, Subsidies, and Standards. *Journal of Public Economic*s, 65(3): 193-205 (July).

Palmer, Karen, Hilary Sigman and Margaret Walls. 1997. "The Cost of Reducing Municipal Solid Waste," *Journal of Environmental Economics and Management* 33: 128-150.

Parks, Richard W. 1977. Determinants of Scrapping Rates for Postwar Vintage Automobiles. *Econometrica.* 45 (5), 1099-1115.

Perwak, Joanne, Muriel Goyer, Leslie Nelken, Edmund Payne, and Douglas Wallace. 1981. *An Exposure and Risk Assessment for Lead*. Washington, DC: U.S. Environmental Protection Agency. EPA Contract Number 68-01-5949.

Pizer, William. 1997. Prices versus Quantities Revisited: The Case of Climate Change. Washington, DC: Resources for the Future Discussion Paper 98-02.

——. 1999. WTO Debacle to Affect Industry. Recycling Laws International 5(6): 1-3. (December).

Rent.net. 2000. Apartments, Rentals and Moving Resources by Rent.Net. http://www.rent.net. (Accessed winter 2000).

RCG/Hagler Bailly, Inc. 1994. *New York State Environmental Externalities Cost Study*.

Salkever, Alex. 1999. Information Age Byproduct: A Growing Trail of Toxic Trash. *The Christian Science Monitor.* December, 1, 1999, p. 1.

Sigman, Hilary. 1995. A Comparison of Public Policies for Lead Recycling. *Rand Journal of Economics* 26: 452-478 (Autumn).

Skipton, Sharon and DeLynn Hay. 1997. Drinking Water: Lead. Cooperative Extension, Institute of Agriculture and Natural Resources, University of Nebraska-Lincoln, Document G97-1333-A. (September 1997). Available at: http://www.ianr.unl.edu/pubs/water/g1333.htm#options. (Accessed February 2001).

Smith, Gerald R. 1999. *Lead Minerals Yearbook.* Washington, D.C.: U.S.

Geological Survey. http://minerals.usgs.gov/minerals/pubs/commodity/lead/380499.pdf (Accessed February 2001).

Solow, Robert. 1959. "Investment and Technical Progress," in *Mathematical Methods in the Social Sciences*, edited by Kenneth Arrow, Samuel Karlin, and Patrick Suppes.(Stanford, California, Stanford University Press), 89-104.

South Carolina State Senate. 1999. Senate Bill 619, (April 19).

Stanford Resources, Inc. 2000. Corporate website at www.stanfordresources.com. (July 15).

Tietenberg, T. H. 1985. *Emissions Trading: An Exercise in Reforming Pollution Policy*, Washington, DC: Resources for the Future.

Townsend, Timothy G., Stephen Musson, Yong-Chul Jang, and Il-Hyun Chung. 1999. “Characterization of Lead Leachability from Cathode Ray Tubes using the Toxicity Characteristic Leaching Procedure.” Florida Center for Solid Waste and Hazardous Management, Report #99-5. (December).

U.S. Bureau of the Census. 1990. *1990 Census of Population and Housing*.

U.S. Bureau of Economic Analysis. 2000. *Final Sales of Computers*.

U.S. Bureau of Labor Statistics. 2000. State and Area Employment, Hours, and Earnings. Data extracted from website (http://www.bls.gov). (Accessed 5 September 2000).

U.S. Environmental Protection Agency. 1986. *Air Quality for Lead Volume II of IV*.

Prepared by the Environmental Criteria and Assessment Office, Research Triangle Park, North Carolina. EPA/600/8-83/028bF.

U.S. Environmental Protection Agency. 1991. *States' Efforts to Promote Lead-Acid Battery Recycling*. Washington, DC: U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency. 1995. *EPA Office of Compliance Sector Notebook Project: Profile of the Electronics and Computer Industry* (EPA/310-R-95-002).

U.S. Environmental Protection Agency. 1998. *Residential Collection of Household End-of-Life Electrical and Electronic Equipment Pilot Collection Project* (EPA-901-R-98-002, February).

U.S. Environmental Protection Agency. 1999. *Analysis of Five Community Consumer/Residential Collections: End of Life Electronic and Electrical Equipment.* (DPA-901-R-98-003, April).

U.S. Environmental Protection Agency. 2000. http://www.epa.gov/epr/products (accessed during January 2000).

University of Massachusetts, Amherst. 1998. "University of Massachusetts Amherst Scrap Electronics Processing – Technical Report #7," Chelsea Center for Recycling and Economic Development (August).

Wade Miller Associates, Inc. and Abt Associates, Inc. 1991. *Final Regulatory Impact*

Analysis of National Primary Drinking Water Regulations for Lead and Copper. Prepared for U.S. Environmental Protection Agency, Office of Drinking Water, Washington, DC. EPA Contract No. 68-CO-0069.

Whelan, Karl. 2000. "Computers, Obsolescence, and Productivity" (Draft, February: Division of Research and Statistics, Federal Reserve Board).

Appendix A: Examples of Programs and Policies

The source of this information is the U.S. Environmental Protection Agency's website (http://www.epa.gov/epr/products), accessed during January 2000, and, in some cases correspondence with local officials. While not comprehensive, this list is intended to illustrate the variety of approaches now underway for managing end-of-life electronics.

Private Programs

- **Sony Electronics –** In October 2000, Sony announced the beginning of a collaboration with the Minnesota Office of Environmental Assistance (MOEA) and Waste Management, Inc. (WMI) to establish a takeback and recycling program for Sony Electronic products (glass from CRTs will be sent to WMI in Pennsylvania for cleaning and then the glass will be sold back to Sony for use in new CRTs). Sony hopes to expand this program to five additional states within the next year and expand the program nationwide within five years.
- **IBM's PC Recycling Service –** In 2000, IBM announced that—for a fee of $29.99—consumers could send used equipment to IBM for recycling or charitable donation.
- **Gateway Country –** The company gives discounts on new computers when people donate an old computer (386 or better) to Goodwill.
- **Dell Computer –** Dell is manufacturing a line of professional computers that are entirely recyclable (OptiPlex PCs).
- **Hewlett Packard –** HP has incorporated design improvements into their products that facilitate disassembly.

Local Programs

Ongoing programs:

- **Hennepin County, MN –** The county began its Residential Consumer Electronics program in fall 1992 and collects materials through drop-off centers, periodic event collection, and municipal collection. The City of Minneapolis collects through curbside and facility collection.

- **Linn County, IA –** The county started an electronics collection program in spring 2000. There are 2 collection sites for the program. Residents will be charged a per-unit processing fee. The county had three pilot programs in 1999 and collected 22.65 tons of material.

Pilot Programs:

- **King County, WA –** The county began a four- month program in early 2000 collecting CPUs, monitors, keyboards, and mice from residents and small to medium-sized businesses. They use collection drop-off sites. Broken monitors that cannot be repaired are stored and then a county-contracted recycler picks them up and takes them to a recycling facility.
- **Franklin County, MA –** The county conducted a one-day collection event in 1998 and, in 1999, conducted two more one-day collections and set up a two month drop-off program. There are plans to develop a permanent drop-off program in 2000 and charge $5 per CRT and $2 for non-CRT equipment to finance the program (the program was originally financed by a grant).
- **Howard County, MD –** The county is conducting a pilot program with a local electronics recycler. The recycling company has placed a roll-off container at the county landfill so residents can drop-off their materials. The recycler accepts the materials at no charge.

One-time collection efforts:

- **Morris County, NJ –** The Morris County Municipal Utilities Authority held a collection of computer equipment (including CRTs) and plans on holding 2 additional special one-day collection events. There is a $5 fee for a monitor and $5 for a CPU.
- **Cuyahoga County, OH –** The county held a computer collection program in August 2000. Inmates in state and federal prisons in Ohio refurbished and upgraded the computers, which were given away or sold at low cost to area schools. Nonworking computers were dismantled for recycling.
- **St. Croix County, WI –** The county held a collection day on June 3, 2000 for computers, TVs and other electronic products. There are plans to refurbish and reuse as much as possible and recycle the remainder.

- **Northern Cook County, IL –** System Service International (SSI, an electronics recycler) and Motorola sponsored two separate collection efforts. SSI disassembled the products to retain the materials that could be used in the production of other products.

State Policies and Programs

- **Florida –** allows CRTs to be managed as materials instead of wastes unless they are being disposed of in landfills or incinerators. The state also is promoting arecycling infrastructure by giving recycling grants to lead recyclers and for collection infrastructure. Florida also is considering a future CRT disposal ban and is executing a state recycling contract**.**
- **Massachusetts –** banned residential consumers and businesses from disposing of CRTs by incineration or landfilling in April 2000. The state helped establish 70 municipal CRT collection programs through its recycling grants program and has established six permanent collection sites throughout the state. Massachusetts is working to promote markets for residential CRTs and to set up a single-payer state recycling contract.
- **Minnesota –** conducted a pilot study with Sony and WMI in 1999. It has recently signed a five-year agreement with Sony to establish an ongoing takeback and recycling program. Hennepin County and the city of Minneapolis have been collecting electronic equipment at the curb since late 1998.
- **Wisconsin –** using funding from the state's Department of Natural Resources to help develop products that contain recycled CRT glass; establishing a nonprofit organization to refurbish and upgrade used computers to sell to schools; working with the Department of Corrections to accept used computers from state government and to train inmates to refurbish computers and demanufacture and recycle end-of-life equipment.
- **Connecticut –** through the State Department of Environmental Protection, is encouraging municipal and regional drop-off sites. Several one-day collections have been held around the state.
- **New Jersey –** Rutgers University has received an EPA grant to encourage the recycling of end-of-life computers and other equipment. The university has produced a video that will be shown to schoolchildren and materials for the children to take home to their parents.

Federal Policies and Programs

In addition to the EPA Common Sense Initiative and the Design for Environment, there are two other programs listed on EPA's website.

- **Computers for Learning –** Program established by Executive Order 12999 "Educational Technology: Ensuring an Opportunity for All Children in the Next Century." This program is designed to facilitate the transfer of surplus federal government equipment to schools and nonprofit organizations.
- **Demanufacturing of Electronic Equipment for Reuse and Recycling (DEER2) –** Program through the National Defense Center for Environmental Excellence of the Department of Defense (DOD). The program uses surplus DOD computer equipment for developing emerging technologies for equipment demanufacturing.

Appendix B: Our Data and Assumptions about the Discard Decision

One of our key assumptions is the age at which monitors are discarded. By discard we mean one of the EOL options we model in the text: storage, recycling (through dismantling and reuse of lead and other component materials), and disposal as waste (landfilling, incineration). As noted, we do not include reuse (for example, through donations to charitable organizations) as a terminal discard option because we assume that it occurs earlier in the lifecycle of the monitor.

The National Safety Council survey (1999) concludes that monitors become technologically obsolete at four years of age and reach their final end of life at six to seven years of age (the council refers to four-year old monitors as having reached the end of their "first life," and six- to seven-year-old monitors as at the end of their "total life"). As a test of the reasonableness of these lifecycle stages, we evaluate the implied relationship between the prices of new and old monitors of different vintages. We collected price data from monitor prices advertised in *The New York Times* business section between 1992 and 1999; the advertised models were chosen to represent monitors of comparable in diameter and color-capability.

We then calculate expression (4) in the text as a test of whether age six as the estimate of $(t\text{-}v)$ seems reasonable. We estimate g from our data on the price of new monitors and use two values of $(\gamma + \delta)$. One value is from research undertaken for the National Income and Product Accounts that reports a preliminary estimate of around 16 % for monitors. The rate of depreciation for computing equipment allowed for U.S. income tax reporting (although different from the economic life remaining for productive use of the asset) assumes a faster depreciation rate—a five-year life for these assets

Our estimated rate of price change is about 16%. With the NIPA value for $(\gamma + \delta)$, we come within about 3% of predicting the price of a new monitor given scrappage of an old one at age 6. With the IRS value—a more generous depreciation rate—we underpredict the price of a new monitor by about 20%.

Appendix C: Description of our Data

Residential Storage Costs

Apartment dwellers

We calculate storage costs for apartment dwellers in our three regions using data obtained from the Internet apartment listing search engine http://www.rent.net (accessed in winter, 2000). We obtained rental rates for December 2000 for cities with listings on the web site for each of the states in our regions. We divided the lowest rental price[27] for a studio and two-bedroom apartment in each state by an approximate size for each type of apartment (studio= 450 sq. ft.; two bedroom= 1,000 sq. ft.) to obtain a rental rate per square foot of apartment space. We then used these rent-per-square-foot values to create a distribution of rental price per square foot for apartment dwellers. We calculate storage cost per monitor by multiplying the rent per square foot by the storage space occupied by a monitor (assumed to be 2 X 2 square feet).

House dwellers

We assume that storage costs for individuals living in a detached house are zero.

Nonresidential Storage Costs

We calculate a distribution of nonresidential storage costs for selected cities in our three regions using rental rates per square foot and estimates of the square feet used for monitor storage. We use 1998 downtown and suburban office rental rates per square foot (CB Richard Ellis National Real Estate Index Market Monitor, 1998). We pooled both the downtown and suburban rates across the three regions to obtain the distribution of rental rates used in the model. We assume that each monitor uses two square feet of storage space and that five monitors may be stacked on top of one another in order to calculate the storage cost per monitor for nonresidential consumers.

[27] These lowest rental rates are a conservative estimate of apartment rental prices in these regions.

Residential Waste Handling Costs

Residential consumers with unit pricing

We estimated the price of waste disposal for the 10% of residential consumers in our model with unit pricing from values obtained in a previous study of household recycling (Jenkins, et al. 2000). This study estimated a mean price of 9.8 cents per gallon and a standard deviation of 3.6 cents per gallon for a sample of households with unit pricing. From those results we assume an average price of 10 cents per gallon for the first bag of trash disposed and an average volume of approximately 30 gallons per monitor to arrive at the cost of waste disposal of $3 per monitor (standard deviation= $1.08) for households with unit pricing.[28]

Residential consumers without unit pricing

We assume that residential consumers without unit pricing of garbage disposal would incur a zero private marginal cost for waste disposal for both the incineration and municipal solid waste landfill options.

Allocation to landfills and incinerators

We use the nationwide percentage of waste sent to landfills and to incinerators in 1998; about 78% and 22%, respectively.

Nonresidential Waste Handling Costs

According to RCRA, there are three classes of waste CRT generators: large quantity generators, small quantity generators and conditionally exempt small quantity generators. Large quantity generators—those that produce more than 1,000 kg of hazardous waste per month (about 85 CRT monitors per year)—must dispose of their CRTs as hazardous waste under RCRA subtitle C. For CRTs, this requirement means there are limitations on storage of used CRTs, record keeping requirements for shipments of used CRTs and that CRTs must be either treated to make them nonhazardous before disposal in a regular landfill or disposed in a hazardous waste disposal facility. Small quantity generators, which produce between 100 and 1000 kg of hazardous waste per month, are subject to a less restrictive set of hazardous waste disposal

[28] We assume this cost of disposal for both incineration and disposal in a MSW landfill.

requirements. Generators that produce less than 100 kg of hazardous waste per month, including CRTs, are exempt from this requirement and may dispose of CRTs as municipal solid waste. Households are assumed to fall into this last category.

Nonresidential consumers subject to RCRA subtitle C

Due to current regulations, these consumers are not legally permitted to dispose of their CRTs as municipal solid waste (thus incineration and disposal in a MSW landfill are not available as options in model). We assume the total cost of hazardous waste disposal for nonresidential consumers ranges from $5.08 to $7.02 per CRT. The hazardous waste processing costs range from $4.48 to $5.77 per CRT, while the transportation costs are between $0.60 to $1.25 per CRT.

Nonresidential consumers not subject to RCRA subtitle C

Nonresidential consumers that are small quantity generators of hazardous waste have the option of disposing of their CRT waste by incineration or landfilling. Total disposal costs include the collection and transportation costs to the landfill or incinerator and the tipping fee paid at the landfill or incinerator. From Ley, Macauley, and Salant, (forthcoming), the estimate of MSW collection and transportation costs ranges from 5 to 11 cents/mile/ton. We use a mean collection and transportation cost of 8 cents/mile/ton which is then multiplied by the average transportation distance. We assume that incinerators and landfills are co-located. In order to obtain an approximate distance to these facilities, we use a sample of facilities in each of our regions, assume the facility is located in the center of the county, and then determine the distance from the center of the county to the county boundary. We use a fitted normal distribution of all distances calculated for our three regions to characterize transportation distance in the model. We obtained average tipping fees for landfills and incinerators for the states in our regions during 1999 to obtain a distribution of tipping fee costs for each type of disposal (Goldstein 2000). We assume each monitor weighs 40 pounds to obtain the total landfill disposal or incineration costs per monitor.

Residential Recycling and Nonresidential Recycling Costs

We assume the drop-off center for recycling is located at the landfill/incineration site, thus we use the distribution of distances described above as the travel distance. We use an estimate of personal travel cost for gasoline and auto wear-and-tear of 0.325 cents per mile (based on the mileage expense allowed by the U.S. tax code). We assume an average traveling time of one hour to travel to and from the drop-off point, which we then multiply by an average wage rate to obtain the opportunity cost of time spent traveling to the drop-off site. We obtained annual average hourly earnings for durable and nondurable goods for 1998 from the Bureau of Labor Statistics for our three sample regions (BLS 1998). We then used the mean and standard deviation of these wages to obtain the distribution of wage rates used in our model. We sum the opportunity cost of travel time and the personal traveling cost to obtain the total drop-off cost.

The costs associated with shipping and processing of CRTs sent directly to a recycling center are from Biddle (2000); ICF Incorporated (1997); and University of Massachusetts (1998). These studies report that the costs of shipping to a recycling center range from $3 to $30 per CRT and demanufacturing and processing costs range from $3 to $15 per unit.

Appendix D: Detailed Model Results

Table D-A1: Predicted EOL Allocation for Policy Scenario A (%)

Ban Incineration and Landfill Disposal

Consumer EOL option	Residential Consumers				Nonresidential Consumers	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	27.10	27.10	99.05	99.05	0.35	77.80
Incineration	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA
Drop off	43.85	43.85	0.90	0.90	NA	NA
Hazardous	NA	NA	NA	NA	99.20	NA
Recycling	29.05	29.05	0.05	0.05	0.45	22.20

Note: NA = Option is not available.

Table D-A2: Predicted Quantity Allocation for Policy Scenario A (Number of CRTs)

Ban Incineration and Landfill Disposal

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	27,830	25,2600	145,500	1,315,000	466	10,250,000	11,99,1396
Incineration	NA	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA	NA
Drop off	45,030	408,800	1,322	11,940	NA	NA	467,092
Hazardous	NA	NA	NA	NA	132,000	NA	132,000
Recycling	29,830	270,800	73	664	599	2,925,000	3,226,966
Total	102,690	932,200	146,895	1,327,604	133,064	13,175,000	15,817,453

Note: NA = Option is not available.

Table D-B1: Predicted EOL Allocation for Policy Scenario B (%)

Ban Disposal and Subsidize Recycling by $10 per CRT

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	7.50	7.50	89.15	89.15	0.30	35.30
Incineration	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA
Drop off	54.70	54.70	8.55	8.55	NA	NA
Hazardous	NA	NA	NA	NA	88.85	NA
Recycling	37.80	37.80	2.30	2.30	10.85	64.70

Note: NA = Option is not available.

Table D-B2: Predicted Quantity Allocation for Policy Scenario B (number of CRTs)

Ban Disposal and Subsidize Recycling by $10 per CRT

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	7,703	69,920	131,000	1,183,000	399	4,652,000	6,044,022
Incineration	NA	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA	NA
Drop off	56,180	509,900	12,560	113,500	NA	NA	692,140
Hazardous	NA	NA	NA	NA	118,200	NA	118,200
Recycling	38,820	352,400	3,380	30,530	14,430	8,526,000	8,965,560
Total	102,703	932,220	146,940	1,327,030	133,029	13,178,000	15,819,922

Note: NA = Option is not available.

Table D-C1: Predicted EOL Allocation for Policy Scenario C (%)

Subsidize Recycling to Achieve 10% Recycling Rate

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	0.90	0.80	77.90	26.12	0.30	0
Incineration	14.66	17.08	0.20	11.49	NA	11.55
Landfill	51.99	60.57	0.25	40.74	NA	80.50
Drop off	22.45	15.45	15.50	15.50	NA	NA
Hazardous	NA	NA	NA	NA	75.45	NA
Recycling	10.00	6.10	6.15	6.15	24.25	7.95

Note: NA = Option is not available.

Table D-C2: Predicted Quantity Allocation for Policy Scenario C (number of CRTs)

Subsidize Recycling to Achieve 10% Recycling Rate

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	924	7,458	114,500	346,700	399	0	469,981
Incineration	15,060	159,200	294	152,500	NA	1,522,000	1,849,054
Landfill	53,390	564,600	367	540,700	NA	10,610,000	11,769,057
Drop off	23,060	144,000	22,780	205,700	NA	NA	395,540
Hazardous	NA	NA	NA	NA	100,400	NA	100,400
Recycling	10,270	56,860	9,037	81,620	32,260	1,048,000	1,238,047
Total	102,704	932,118	146,978	1,327,220	133,059	13,180,000	15,822,080

Note: NA = Option is not available.

Table D-D1: Predicted EOL Allocation for Policy Scenario D (%)

Subsidize Recycling to Achieve 23% Recycling Rate

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	0.65	0.60	57.10	19.15	0.30	0
Incineration	9.82	12.53	0.09	8.43	NA	10.30
Landfill	34.83	44.42	0.31	29.87	NA	70.55
Drop off	35.25	28.55	28.55	28.60	NA	NA
Hazardous	NA	NA	NA	NA	56.25	NA
Recycling	19.45	13.90	13.95	13.95	43.45	19.15

Note: NA = Option is not available.

Table D-D2: Predicted Quantity Allocation for Policy Scenario D (number of CRTs)

Subsidize Recycling to Achieve 23% Recycling Rate

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	668	5,593	83,900	254,200	399	0	344,760
Incineration	10,090	116,800	132	111,900	NA	1,357,000	1,595,922
Landfill	35,770	414,100	456	396,400	NA	9,297,000	10,143,726
Drop off	36,200	266,100	41,950	379,600	NA	NA	723,850
Hazardous	NA	NA	NA	NA	74,830	NA	74,830
Recycling	19,980	129,600	20,500	185,100	57,800	2,523,000	2,935,980
Total	102,708	932,193	146,938	1,327,200	133,029	13,177,000	15,819,067

Note: NA = Option is not available.

Table D-E1: Predicted EOL Allocation for Policy Scenario E (%)

Ban Incineration Only

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	6.61	6.34	98.70	48.05	0.35	9.89
Incineration	NA	NA	NA	NA	NA	NA
Landfill	75.63	76.60	0.35	51.49	NA	87.29
Drop off	10.69	10.26	0.90	0.44	NA	NA
Hazardous	NA	NA	NA	NA	99.20	NA
Recycling	7.08	6.80	0.05	0.02	0.45	2.82

Note: NA = Option is not available.

Table D-E2: Predicted Quantity Allocation for Policy Scenario E (number of CRTs)

Ban Incineration Only

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	6,788	59,100	145,000	637,700	466	1,303,000	2,152,054
Incineration	NA	NA	NA	NA	NA	NA	NA
Landfill	77,670	714,100	514	683,400	NA	11,500,000	12,975,684
Drop off	10,980	95,640	1,322	5,840	NA	NA	113,782
Hazardous	NA	NA	NA	NA	132,000	NA	132,000
Recycling	7,271	63,390	73	265	599	371,600	443,199
Total	102,709	932,230	146,910	1,327,205	133,064	13,174,600	15,816,718

Note: NA = Option is not available.

Table D-F1: Predicted EOL Allocation for Policy Scenario F (%)

Curbside Collection without Disposal Ban

Consumer EOL option	Residential Consumers				Nonresidential Consumers	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	0.85	0.85	49.25	24.76	0.35	NA
Incineration	0.20	14.40	0.12	10.90	NA	12.70
Landfill	0.35	51.06	0.43	38.63	NA	87.25
Drop off	0.90	0.90	0.90	0.90	NA	NA
Curbside	97.65	32.73	49.25	24.76	NA	NA
Hazardous	NA	NA	NA	NA	99.20	NA
Recycling	0.05	0.05	0.05	0.05	0.45	0.05

Note: NA = Option is not available.

Table D-F2: Predicted Quantity Allocation for Policy Scenario F (number of CRTs)

Curbside Collection without Disposal Ban

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**		**Total**
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	873	7,924	72,370	328,600	466	NA	410,233
Incineration	205	134,200	176	144,700	NA	1,674,000	1,953,282
Landfill	360	476,000	632	512,700	NA	11,500,000	12,489,691
Drop off	924	8,390	1,322	11940	NA	NA	22,576
Curbside	100,300	305,100	72,370	328,600	NA	NA	806,370
Hazardous	NA	NA	NA	NA	132,000	NA	132,000
Recycling	51	466	73	664	599	6,589	8,442
-Total	102,714	932,080	146,944	1,327,204	133,064	13,180,589	15,822,594

Note: NA = Option is not available.

Table D-G1: Predicted EOL Allocation for Policy Scenario G (%)

Curbside Collection with Disposal Ban

Consumer EOL option	**Residential Consumers**				**Nonresidential Consumers**	
	Apartment dwellers		House dwellers			
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators
Storage	0.85	0.85	49.53	49.53	0.35	77.80
Incineration	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA
Drop off	0.90	0.90	0.90	0.90	NA	NA
Curbside	98.20	98.20	49.53	49.53	NA	NA
Hazardous	NA	NA	NA	NA	99.20	NA
Recycling	0.05	0.05	0.05	0.05	0.45	22.20

Note: NA = Option is not available.

Table D-G2: Predicted Quantity Allocation for Policy Scenario G (number of CRTs)

Curbside Collection with Disposal Ban

Consumer EOL option	Residential Consumers				Nonresidential Consumers		Total
	Apartment dwellers		House dwellers				
	with unit pricing	without unit pricing	with unit pricing	without unit pricing	Hazardous waste generators	Non-hazardous waste generators	
Storage	873	7,924	72,780	657,400	466	10,250,000	10,989,443
Incineration	NA	NA	NA	NA	NA	NA	NA
Landfill	NA	NA	NA	NA	NA	NA	NA
Drop off	924	8,390	1,322	11,940	NA	NA	22,576
Curbside	100,900	915,400	72,780	657,400	NA	NA	1,746,480
Hazardous	NA	NA	NA	NA	132,000	NA	132,000
Recycling	51	466	73	664	599	2,925,000	2,926,853
Total	102,749	932,180	146,955	1,327,404	133,064	13,175,000	15,817,352

Note: NA = Option is not available.

Appendix E: Using Tradable Recycling Credits to Implement Producer Take Back for CRTs

As discussed in our paper, an increasingly popular policy for promoting the recycling of particular products or materials is to require producers to take back these items from consumers at the end of their lives and to make producers responsible for recycling and ultimate disposal. These programs usually combine the take back requirement with a minimum recycling rate requirement. Such programs are particularly popular in Europe where Germany was among the first countries to require manufacturers to take back product packaging and where the E.U. is currently considering a proposal to require manufacturer take-back of used electronics which would include CRTs. Takeback programs are growing in popularity throughout the world, as evidenced by the Japanese program to require takeback of used household appliances such as televisions and refrigerators.

One issue that has arisen in the debate over the E.U. Directive on Waste Electronic Equipment is whether the responsibility to achieve the recovery and recycling rate targets should be assigned to specific firms or spread across the entire industry. Firms that have made significant progress in making their products more recyclable favor the former approach while firms that have not done so tend to favor the latter. If the goal of the policy is to obtain a particular level of product recovery (in Europe, this generally includes energy recovery through incineration as well as recycling) or recycling, then forcing all firms to achieve the same recycling rate may not be the most efficient way to achieve the overall goal, particularly if some firms' products are more expensive to recycle or to make more recyclable than others. Such an inflexible policy could be quite costly, particularly if there are wide differences among products in terms of ease of recycling. However, having an industry-wide target makes it possible for some firms to free-ride on the prior efforts of others. Also, such a policy may be difficult to enforce without a clear delineation of which entity is responsible for doing what.

A possible solution to the problems inherent in both of these approaches would be to establish a system of tradable CRT recycling credits. Economic theory suggests that allowing sources to trade the right to pollute will achieve the desired environmental goal at least cost (Tietenberg 1985). For an environmental problem such as air pollution, a tradable allowance system generally works by capping total emissions of the pollutant of concern and then allowing sources of that pollution to trade the right to emit up to the aggregate cap. Firms with a relatively low cost of reducing pollution will be able to sell pollution allowances to firms with relatively high pollution control costs.

In the case of CRT recycling, the tradable commodity would be recycling credits.[29] Policymakers would set a target recycling rate for EOL CRTs. Each time a CRT is collected for recycling and recycled, the manufacturer of that CRT receives a recycling credit issued by the government.[30] At the end of the year, all manufacturers are required to give to the government recycling credits equal to a certain percentage of the CRTs that they sold that year (or perhaps in a prior year that better reflects the pool of EOL CRTs currently becoming obsolete). If a manufacturer fails to recycle the minimum percentage, she has the opportunity to purchase a recycling credit from another manufacturer that has recycled more than the required amount. Manufacturers who have relatively low costs of recycling will have an incentive to over comply with the minimum recycling rate goal because they can sell their excess recycling credits to other CRT producers. Manufacturers who have high costs of recycling can use the credit market to purchase the recycling credits necessary to satisfy their obligation under the regulation. By establishing property rights in these recycling credits, this system reduces the potential for free riding while still allowing for the flexibility inherent in an industry-wide standard that does not exist with a firm-specific standard.

Tradable credit programs also provide incentives for firms to set up collection programs and make their products more recyclable. In an effort to generate needed recycling credits, firms will seek cost-effective ways to collect used CRTs from households and businesses. Also, because the manufacturer gets a credit each time one of its CRTs is recycled, she will have an incentive to make product design changes that reduce the cost of ultimately recycling the product.

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[29] Tradable credit systems have been proposed in the United States to encourage the use of renewables technologies to generate electricity. These systems require that a certain minimum percentage of the electricity generated in a particular year must be produced with renewable technologies. Renewable generators received credits for each mega-watt hour of power they generate and these credits can be traded. For more information see Clemmer, Nogee and Brower (1999).

[30] This mechanism assumes that CRTs are labeled in such a way that it is easy to know who manufactured the CRT and, therefore, should receive the credit.

Designing a tradable credit program to promote CRT recycling is a complicated undertaking. Several features of such a program need to be specified, including the basis for the recycling rate goal, the allocation mechanism for the credits, how the level of the aggregate recycling rate goal changes over time, rules governing inter-temporal trading of credits, whether or not to provide for a price cap on credits and how orphan products will be handled. We briefly discuss each of these features in turn below. To fully address each of these issues requires more research.

Defining the basis of the recycling rate goal

As described above, this tradable credit program is designed to achieve a particular recycling rate goal at least cost. Because CRTs are a durable product, there is a substantial period of time (four to seven years) between the original sale and the product's EOL. The basis for the recycling rate is ideally the number of CRTs that have reached the end of their useful lives at a given point in time. The relationship between the size of this group and the number of CRTs sold in the current year depends on how much the market has grown and the rate of obsolescence. If the market is growing it would not be reasonable to base the recycling rate requirement on current sales. A more relevant basis might be past sales.

Allocating the credits

Under this program, producers are those who would be responsible for handing credits over to the government at the end of the year equal to the required fraction of the relevant basis for their product (presumably past CRT production). In theory, these credits could be allocated to producers, recyclers, or even consumers and then the producer could purchase the credits they needed from whoever held them. Indeed, there might be some merit in allocating credits to recyclers, since they are the ones doing the actual recycling and such an allocation scheme could reduce monitoring costs for the program. However, allocating permits to the recyclers will probably provide weak incentives to the producers to modify the designs of their products to make them more recyclable, since they won't be able to capture much of the benefits of doing so. In this case, making a more recyclable product will reduce the price of recycling credits paid by everyone, and it will slightly reduce the cost of credits the manufacturer will have to bear. On the other hand, if the credits were allocated to the manufacturer, then by making her products more recyclable she would be able to satisfy her requirement with her own permits and then would be able to keep the benefits.

Adjusting the recycling rate goal over time

A tradable-credits program also could be used to smooth the path toward achieving an ambitious long-term recycling rate goal.[31] One complaint about the E.U. Electronics Waste Directive is that the target recycling rates are too high to be achievable in the near term. To allow for the development of recycling technologies and infrastructure, and to reduce the immediate cost impact of the program, the aggregate recycling rate target could be ratcheted up over time and manufacturers could be allowed to bank credits that were created in early years for use in later years. For example, if a firm recycled more than the relatively low target during an early year, it could either sell its excess credits to other firms for use in that year or save the credits to be applied against the more stringent target in future years. This feature would provide incentives for early increases in recycling activity and at the same time would limit the high costs of complying with the program.

Capping the credit price

In the debate over the E.U. Directive on Electronics Waste, there has been much discussion about the potentially high costs of this program. One way to avoid high costs would be to impose a cap on the price of a credit.[32] Under such a program, the government would cap the price of tradable credits. If the market price rose to the level of the cap, then producers would be able to purchase "paper" credits from the government—at the cap—up to their required level. The revenues from the sale could be used to subsidize CRT recycling or to displace other taxes.

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[31] The sulfur dioxide trading program established under Title IV of the 1990 Clean Air Act Amendments had such a provision. The program was implemented in two phases. In the first phase, emissions of the dirtiest generating units were capped. In the second phase, which began five years into the program, the cap was extended to cover all generators and it was tightened considerably. Plants that over complied in the first period of the program were allowed to bank emission allowances for use in the second period. The removal of lead from gasoline in the United States was achieved under a regulatory program that imposed a decreasing lead content standard for leaded gasoline in the 1980s and that allowed trading of lead rights among refineries and banking of lead rights for use in future time periods. Under the trading program, lead content in gasoline was reduced from 1.1 grams per gallon to 0.1 grams. After the trading program ended, the standard was subsequently reduced to 0 in the mid 1990s.

[32] For an example of such a proposal see Pizer (1997).

Orphan products

An important issue for implementing this program is what to do about orphan products. These are CRTs where manufacturers have gone out of business. Important questions here are who would be responsible for recycling these products and who would get the credits when they actually are recycled.

Two potential drawbacks of the tradable credits approach are worth noting here. First, there would be administrative costs and transactions costs associated with setting up and operating such a system. The assumption here is that if those costs were higher for this system than under a more traditional producer takeback system (and it's not necessarily the case they would be higher), the cost savings associated with a more flexible mechanism would offset the higher administrative and transaction costs. Whether or not this is the case is a subject for future research.

Second, the credit trading system proposed here represents a potentially more efficient alternative to traditional takeback programs for achieving a desired recycling-rate goal. However, prior research (Palmer and Walls 1997) tells us that a recycling rate target is not a socially efficient environmental policy. If the concern associated with CRT disposal is primarily related to releases of lead into the environment, then encouraging recycling is not the most efficient way to address that concern because policies that target recycling do not encourage an efficient level of source reduction. Instead, policies should target directly releases of lead into the environment. An example of such a policy would be a deposit-refund that was tied to the lead content of a CRT. Such a policy would impose a deposit on CRT purchases and provide a refund when CRTs are returned for recycling. The deposit/refund would vary with the lead content of the CRT, which presumably depends on the size of the monitor. Depending on the size of the deposit/refund this instrument would provide incentives for substituting smaller CRTs for large CRTs, extending the life of existing CRTs, and increased recycling of CRTs.