REVISED DRAFT

HAZARD ASSESSMENT OF

PERFLUOROOCTANOIC ACID

AND ITS SALTS

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Office of Pollution Prevention and Toxics
Risk Assessment Division

November 4, 2002

PREFACE

This is a preliminary assessment of the potential hazards to human health and the environment associated with exposure to perfluorooctanoic acid (PFOA) and its salts. The majority of the toxicology information is for ammonium perfluorooctanoic acid (APFO). This assessment includes a review of the studies that were available as of October, 2002.

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EXECUTIVE SUMMARY

Introduction

Perfluorooctanoic acid (PFOA) and its salts are fully fluorinated organic compounds that can be produced synthetically or through the degradation or metabolism of other fluorochemical products. PFOA is primarily used as a reactive intermediate, while its salts are used as processing aids in the production of fluoropolymers and fluoroelastomers and in other surfactant uses. In recent years, less than 600 metric tons per year of PFOA and its salts have been manufactured in the United States or imported. Most of the toxicology studies have been conducted with the ammonium salt of perfluorooctanoic acid, which is referred to as APFO in this report.

Environmental Fate and Effects

PFOA is persistent in the environment. It does not hydrolyze, photolyze or biodegrade under environmental conditions.

Groundwater samples taken near fire-training areas that used fire-fighting foams containing perfluorinated surfactants had elevated PFOA concentrations many years after the foam use. This demonstrates the following: (1) PFOA either existed in--or was formed via degradation of--the surfactants, (2) PFOA or its precursors migrate through the soil, and (3) PFOA persists in groundwater.

Several wildlife species have been sampled around the world to determine levels of PFOA. PFOA has rarely been found in fish sampled from the U.S., certain European countries, the North Pacific Ocean and Antarctic locations, or in fish-eating bird samples collected from the U.S., including Midway atoll, the Baltic and Mediterranean Seas, and Japanese and Korean coasts. PFOA was found in a few mink livers from Massachusetts, but not found in mink from Louisiana, South Carolina and Illinois. PFOA concentrations in river otter livers from Washington and Oregon States were less than the quantification limit of 36 ng/g, wet wt. PFOA was not detected at quantifiable concentrations in oysters collected in the Chesapeake Bay and Gulf of Mexico of the U.S. coast.

The concentrations of PFOA in surface water, sediments, clams, and fish collected from locations upstream and downstream of the 3M manufacturing facility at Decatur AL have been determined. Of the three downstream water and sediment sampling locations, the two closest to the facility had PFOA surface water concentrations significantly greater than the two upstream sites; the three downstream locations also had sediment concentrations significantly greater than the upstream sites. The small sample size prevented determination of significance for fish whole body PFOA concentrations. The average PFOA concentration in clams was not significantly different between the upstream and downstream locations.

Based on available laboratory data, APFO does not appear to bioaccumulate in fish. In a study of fathead minnows, the calculated BCF for APFO was 1.8. In a study of carp, the BCF ranged

from 3.1 to 9.1.

Several species were tested to assess the acute toxicity of APFO; these included the fathead minnow (*Pimephales promelas*), bluegill sunfish (*Lepomis machrochirus*), water flea (*Daphnia magna*), and a green algae (*Selenastrum capricornutum*). Comparisons of the different studies are problematic for several reasons. The studies were conducted with different test substances. Generally the ammonium salt or the tetrabutylammonium salt was tested. Purity of the test material is a major concern and was not sufficiently characterized in these tests. In some tests it appeared that 100% test chemical was used, for others a chemical of lesser purity (approximately 27 to 85%) was used. Water, a solvent (isopropanol) or a combination of both was used in other tests, for no obvious stated reason. Finally, only nominal test chemical concentrations were reported; the actual concentrations were not reported.

Twelve tests were conducted with fathead minnows; 96-h LC50 values (based on mortality) ranged from 70 to 843 mg/L. It is unclear why this range is so wide. Assuming these studies are valid, and due to the limitations discussed above, these toxicity values indicate low toxicity. The two acute values for bluegill sunfish also indicate low toxicity (96-h LC50s of >420, and 569 mg/L).

Nine acute tests were conducted with daphnids and 48-h EC50 values (based on immobilization) ranged from 39 to >1000 mg/L. The lower values are indicative of moderate toxicity, but the wide range makes interpretation difficult.

Seven tests were conducted with green algae; 96-h EC50 values (based on growth rate, cell density, cell counts, and dry weights) ranged from 1.2 to >666 mg/L (the EC50 cell density value of 1,000 mg/L is excluded from this discussion). The lower value indicates high to moderate toxicity, based on the acute criteria. The lower value would also be indicative of moderate toxicity, based on the chronic moderate criterion ($0.1 \leq 10$ mg/L). A 14-d EC50 value of 43 mg/L, based on cell counts, for green algae was also calculated in one study. This is indicative of low chronic toxicity, based on the chronic criterion (10 mg/L). Green algae appeared to be the most sensitive test species in the 44% APFO test sample, daphnids were the next most sensitive, and fathead minnows were the least sensitive.

Human Health Effects and Biomonitoring

Little information is available concerning the pharmacokinetics of APFO in humans. An ongoing 5-year, half-life study in 9 retired workers has suggested a mean serum PFOA half-life of 4.37 years (range, 1.50 – 13.49 years). These data provide evidence of the potential to bioaccumulate PFOA in humans.

Animal studies have shown that APFO is well absorbed following oral and inhalation exposure, and to a lesser extent following dermal exposure. In the past, Chemolite workers have been exposed to large dermal doses of PFOA. It appears that dermal exposure may have played a significant role in the absorption of PFOA in these workers. Upon recognition that PFOA could

be absorbed dermally, work practices were changed and engineering controls were adopted that reduced dermal exposures.

PFOA distributes primarily to the liver, plasma, and kidney, and to a lesser extent, other tissues of the body including the testis and ovary. It does not partition to the lipid fraction or adipose tissue. PFOA binds to macro-molecules in the tissues listed above. PFOA is not metabolized and there is evidence of enterohepatic circulation of the compound. The urine is the major route of excretion of PFOA in the female rat, while the urine and feces are both major routes of excretion in male rats. There are major gender differences in the elimination of PFOA in rats. In female rats, the half-life is 24 h in the serum and 60 h in the liver; in male rats, the half-life is 105 h in the serum and 210 h in the liver. The rapid excretion of PFOA by female rats is due to active renal tubular secretion (organic acid transport system); this renal tubular secretion is believed to be hormonally controlled, since castrated male rats treated with estradiol have excretion rates of PFOA similar to those of female rats. Hormonal changes during pregnancy do not appear to change the rate of elimination in rats. This gender difference has not been observed in primates and humans.

There are limited data on PFOA serum levels in workers and the general population. Occupational data from plants in the U.S. and Belgium that manufacture or use PFOA indicate that the most recent mean serum levels in workers range from 0.84 to 6.4 ppm. The highest level reported in a worker in 1997 was 81.3 ppm. In non-occupational populations, serum PFOA levels were much lower. In both pooled blood bank samples and in individual samples, mean serum PFOA levels ranged from 3 to 17 ppb. The highest serum PFOA levels were reported in a sample of children from different geographic regions in the U.S. (range, 1.9 – 56.1 ppb).

Epidemiological studies on the effects of PFOA in humans have been conducted on workers. Two mortality studies, a morbidity study, and studies examining effects on the liver, pancreas, endocrine system, and lipid metabolism, have been conducted to date. In addition, a cross-sectional as well as a longitudinal study of the worker surveillance data have recently been submitted. However, these latest 2 studies focus primarily on PFOS rather than PFOA, even though recent PFOA levels are similar to or higher than PFOS levels in workers at these plants. (It should be noted that PFOS levels in the sampled general population are much higher than PFOA levels).

A retrospective cohort mortality study demonstrated a weak, although statistically significant association between prostate cancer mortality and employment duration in the chemical facility of a plant that manufactures PFOA. However, in a recent update to this study in which more specific exposure measures were used, a significant association for prostate cancer was not observed. In a morbidity study, workers with the highest PFOA exposures for the longest durations sought care more often for prostate cancer treatment than workers with lower exposures.

Another study reported an increase in estradiol levels in workers with the highest PFOA serum levels; however, none of the other hormone levels analyzed indicated any adverse effects. Some

of the same employees who participated in the hormone study also were included in a study of cholecystokinin (CCK) levels in employees. No positive association was noted between CCK values and PFOA. The other available study examined cholesterol and other serum components in workers. There did not appear to be any significant differences among workers of different exposure levels. At plants where the serum PFOA levels were lower, cross-sectional and longitudinal studies found positive significant associations between PFOA and cholesterol and triglyceride levels. In addition, a positive, significant association was reported between PFOA and T3 hormone and a negative association with HDL in the cross-sectional study. There are many limitations to these studies, and therefore, all of these results must be interpreted carefully.

In acute toxicity studies in animals, the oral LD50 values for CD rats were >500 mg/kg for males and 250-500 mg/kg for females, and <1000 mg/kg for male and female Wistar rats. There was no mortality following inhalation exposure of 18.6 mg/L for one hour in rats. The dermal LD50 in rabbits was determined to be greater than 2000 mg/kg. APFO is a primary ocular irritant in rabbits, while the data regarding potential skin irritancy are conflicting.

APFO is not mutagenic. APFO did not induce mutation in either S. typhimurium or E. coli when tested either with or without mammalian activation. APFO did not induce gene mutation when tested with or without metabolic activation in the K-1 line of Chinese hamster ovary (CHO) cells in culture. APFO did not induce chromosomal aberrations in human lymphocytes when tested with and without metabolic activation up to cytotoxic concentrations. APFO was tested twice for its ability to induce chromosomal aberrations in CHO cells. In the first assay, APFO induced both chromosomal aberrations and polyploidy in both the presence and absence of metabolic activation. In the second assay, no significant increases in chromosomal aberrations were observed without activation. However, when tested with metabolic activation, APFO induced significant increases in chromosomal aberrations and in polyploidy. APFO was negative in a cell transformation assay in C_3H $10T_{½}$ mouse embryo fibroblasts and in the mouse micronucleus assay.

Subchronic studies in rats and mice with 28 and 90-days of exposure have demonstrated that the liver is the primary target organ. In rat studies, males are far more sensitive than females. Dietary exposure to APFO for 90 days resulted in significant increases in liver weight and hepatocellular hypertrophy in female rats at 1000 ppm (76.5 mg/kg/day) and in male rats at doses as low as 100 ppm (5 mg/kg/day). Analyses of serum and liver levels of PFOA showed a marked gender difference that accounts for the difference in sensitivity. Chronic dietary exposure of rats to 300 ppm APFO (14.2 and 16.1 mg/kg/day for males and females, respectively) for 2 years resulted in increased liver and kidney weights, hematological effects and liver lesions in males and females. In addition, testicular masses were observed in males at 300 ppm and ovarian tubular hyperplasia was observed in females after exposure to 30 ppm (1.6 mg/kg/day), the lowest dose tested.

In a 90-day study with rhesus monkeys, exposure to doses of 30 mg/kg/day or higher resulted in death, lipid depletion in the adrenals, hypocellularity of the bone marrow, and moderate atrophy of the lymphoid follicles in the spleen and lymph nodes. Unlike rodent studies, analyses of the

serum and liver levels did not reveal a gender difference in monkeys, but the sample size was very small (N=2). In a 6-month study of male cynomolgus monkeys, dosing of animals in the 30 mg/kg/day dose group was stopped from days 11–21 because of toxicity. When dosing was resumed on day 22, animals received 20 mg/kg/day and this group was designated the 30/20 mg/kg/day group. This treatment was also not tolerated and treatment was stopped for 3/6 monkeys. Mortality was observed in one monkey at 3 mg/kg/day and at 30/20 mg/kg/day. There were no consistent effects on hormone levels. Increased absolute and relative liver weights were noted at 3, 10 and 30/20 mg/kg/day. While there was no evidence of peroxisome proliferation, there was evidence of mitochondrial proliferation suggesting a different mode of action than observed in rats. The serum levels were highly variable and should be treated with caution. On day 9 of treatment, the serum levels were 126 ± 36.1 µg/mL in the 3 mg/kg/day group and 1597 ± 2392 µg/mL in the 30/20 mg/kg/day group, and during weeks 26/27, the serum levels were 52.5 ± 9.14 µg/mL in the 3 mg/kg/day group and 51.5 ± 77.6 µg/mL in the 30/20 mg/kg/day group. The LOAEL for this study was 3 mg/kg/day and a NOAEL was not established.

PFOA is immunotoxic in mice. Feeding C57Bl/6 mice a diet containing 0.02% PFOA resulted in adverse effects to both the thymus and spleen. In addition, this feeding regimen resulted in suppression of the specific humoral immune response to horse red blood cells, and suppression of splenic lymphocyte proliferation in response to LPS and ConA. The suppressed mice recovered their ability to generate a humoral immune response when they were fed a diet devoid of PFOA. Studies using transgenic mice showed that the peroxisome proliferator-activated receptor alpha was involved in causing the adverse effects to the immune system.

Prenatal developmental toxicity studies in rats resulted in death and reduced body weight in dams exposed to oral doses of 100 mg/kg/day or by inhalation to 25 mg/m^3 APFO. There was no evidence of developmental toxicity after oral exposure to doses as high as 150 mg/kg/day, while inhalation exposure to 25 mg/m^3 resulted in reduced fetal body weights. In a rabbit oral developmental toxicity study there was a significant increase in skeletal variations after exposure to 50 mg/kg/day APFO. There was no evidence of maternal toxicity at 50 mg/kg/day, the highest dose tested.

In a two-generation reproductive toxicity study in rats exposed to 0, 1, 3, 10, and 30 mg/kg/day APFO, significant increases in absolute and relative liver and kidney weights were observed in F0 males at 1 mg/kg/day, while significant reductions in absolute and relative kidney weights were observed in F0 females at 30 mg/kg/day. Reproductive indices were not affected in the F0 animals. Serum levels of the 10 and 30 mg/kg/day groups were measured for F0 males after mating and F0 females at weaning of the F1 pups. In F0 males, the serum levels were 51.1$\pm$9.30 and 45.3$\pm$12.6 ug/l, respectively for the 10 and 30 mg/kg/day groups, and in F0 females, the serum levels were 0.37$\pm$0.0805 and 1.02$\pm$0.425 ug/l, respectively for the 10 and 30 mg/kg/day groups. In F1 females, there was a significant increase in post weaning mortality, a significant decrease in mean body weight, and a significant delay in sexual maturation at 30 mg/kg/day. In F1 males, significant decreases in body weights and body weight gains, and significant changes in absolute liver and spleen weights and in the ratios of liver, kidney, and spleen weights-to-

brain weights were observed in all treated groups. The increase in post weaning mortality and the delay in sexual maturation was also noted in F1 males at 30 mg/kg/day. Reproductive indices were not affected in the F1 animals. The LOAEL for the F1 females was 30 mg/kg/day, and the NOAEL was 10 mg/kg/day; the LOAEL for F1 males was 1 mg/kg/day and a NOAEL was not determined. The difference in sensitivity is presumed to be related to the gender difference in elimination of APFO. No treatment-related effects were observed in the F2 generation. However, it should be noted that the F2 pups were sacrificed at weaning, and thus it was not possible to ascertain the potential post-weaning effects that were noted in the F1 generation.

Carcinogenicity studies in Sprague-Dawley (CD) rats show that APFO is weakly carcinogenic, inducing Leydig cell adenomas in the male rats and mammary fibroadenomas in the females following dietary exposure to 300 ppm for 2 years (equivalent to 14.2 mg/kg/day in males and 16.1 mg/kg/day in females). APFO has also been reported to be carcinogenic toward the liver and pancreas of male CD rats at 300 ppm.

The mechanism(s) of APFO tumorigenesis is not clearly understood. Available data appear to indicate that the induction of tumors by APFO is due to a non-genotoxic mechanism, involving activation of receptors and perturbations of the endocrine system. There is sufficient evidence to suggest that APFO is a PPARα-agonist and that the liver carcinogenicity/toxicity of APFO is mediated by binding to PPARα in the liver. Recently, IARC (1995) concluded that the liver tumors induced in rodents by PPARα-agonists are unlikely to be operative in humans based on our current understanding of the animal mode of action. The Agency is currently examining the scientific knowledge associated with PPARα-agonist–induced liver tumors in rodents and the relevance to humans. Available data suggest that the induction of Leydig cell tumors (LCT) and mammary gland neoplasms by APFO may be due to hormonal imbalance resulting from activation of the PPARα and induction of the cytochrome P450 enzyme, aromatase. Preliminary data suggest that the pancreatic acinar cell tumors are related to an increase in serum level of the growth factor, cholecystokinin.

1.0 Chemical Identity

Chemical Name: Perfluorooctanoic Acid
Molecular formula: C8 H F15 O2

Structural formula: F-CF2-CF2-CF2-CF2-CF2-CF2-CF2-C(=O)-X,

The free acid and some common derivatives have the following CAS numbers:
The perfluorooctanoate anion does not have a specific CAS number.

Free Acid	(X = OM+; M = H)	[335-67-1]
Ammonium Salt	(X = OM+; M = NH4)	[3825-26-1]
Sodium Salt	(X = OM+; M = Na)	[335-95-5]
Potassium Salt	(X = OM+; M = K)	[2395-00-8]
Silver Salt	(X = OM+; M = Ag)	[335-93-3]
Acid Fluoride	(X = F)	[335-66-0]
Methyl Ester	(X = CH3)	[376-27-2]
Ethyl Ester	(X = CH2-CH3)	[3108-24-5]

Synonyms: 1-Octanoic acid, 2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-pentadecafluoro-
PFOA

1.1 Physicochemical Properties

For this report, perfluorooctanoic acid is consistently referred to as PFOA. Most of the toxicology studies have been conducted with the ammonium salt of perfluorooctanoic acid, which will be referred to as APFO in this report. PFOA is a completely fluorinated organic acid. The typical structure has a linear chain of eight carbon atoms. The physical chemical properties noted below are for the free acid, unless otherwise stated. The data for the free acid, pentadecafluorooctanoic acid [335-67-1], is the most complete. The reported vapor pressure of 10 mm Hg appears high for a low melting solid when compared to other low melting solids (chloroacetic acid: solid; MP = 61 to 63 °C; BP = 189 °C; VP = 0.1 kPa (0.75 mm Hg) @ 20 °C; NIOSH), but is consistent with other perfluorinated compounds with similar boiling points (perfluorobutanoic acid BP = 120 °C, VP 10 mm Hg @ 20 °C; Beilstein IV 2 p. 811). Another explanation may be that the 10 mm vapor pressure was measured at an elevated temperature (but the temperature inadvertently omitted), as perfluorooctanoic acid is typically handled as a liquid at 65 °C (3M data sheet for FC-26).

The free acid is expected to completely dissociate in water, leaving the anionic carboxylate in the water and the perfluoroalkyl chain on the surface. In aqueous solutions, individual molecules of PFOA anion loosely associate on the water surface and partition between the air / water

interface. Several reports note that PFOA salts self-associate at the surface, but with agitation they disperse and micelles form at higher concentrations. (Simister, 1992; Calfours, 1985; Edwards, 1997). Water solubility has been reported for PFOA, but it is unclear whether these values are for a microdispersion of micelles, rather than true solubility. Due to these same surface-active properties of PFOA, and the test protocol for the OECD shake flask method, PFOA is anticipated to form multiple layers in octanol/water, much like those observed for PFOS. Therefore, an n-octanol/water partition coefficient cannot be determined.

The available physicochemical properties for the PFOA free acid are:

MW: 414 (Beilstein, 1975)
MP: 45 – 5 °C (Beilstein, 1975)
BP: 189 – 192 °C / 736 mm Hg (Beilstein, 1975)
VP: 10 mm Hg @ 25 °C (approx.) (Exfluor MSDS)
Sol. - Water: 3.4 g/L (telomeric [MP = 34 °C ref. 0.01 - 0.02 mol/L ~4 - 8 g/L) (MSDS from Merck, Fischer, and Chinameilan Internet sites)
pKa: 2.5 (USEPA AR226- 0473)
pH (1g/L): 2.6 (MSDS Merck)

The PFOA derivative of greatest concern and most wide spread use is the ammonium salt (APFO), [3825-26-1]). This substance was the testing substance for the toxicity testing performed and has the P-Chem data provided in the Table below.

The water solubility of APFO has been inconsistently reported. One 3M study reported the water solubility of APFO to be > 10%. It was noted in an earlier study that at concentrations of 20 g/L, the solution "gelled" (3M, 1979). These numbers seem surprising low for a salt , in light of Apollo Scientific selling a 31% aq. solution of APFO. One author reported the APFO partition coefficient log Pow = 5. Another author reported an estimated APFO log Pow = -0.9. This value might not be accurate due to the estimation method used (Hansch and Leo 1979). Again, the anticipated formation of an emulsified layer between the octanol and water surface interface would make determination of log Kow impossible.

Determination of the vapor pressure of APFO is complicated. APFO had recently reported a vapor pressure of 7 x 10-5 mm Hg at 20E C, which seems too low for a material that sublimates as the ammonium salt. (3M Environmental Laboratory, 1993). The ammonium salt begins to sublimate at 130 °C (USEPA AR-226 473). As the temperature increases from when APFO begins to sublimate, 20% of the sample weight is lost by 169 °C. Other salts (Cs, K, Ag, Pb, Li) do not demonstrate similar weight loss until 237 °C or higher. (Lines, 1984). Decomposition of different salts produces perfluoroheptene (loss of metal fluoride and carbon dioxide). This occurs at 320 °C for the sodium salt and at 250-290 °C for the silver salt (Beilstein 1975).

The physicochemical properties of PFOA and its common derivatives are summarized in Table 1.

Table 1. Reported Physicochemical Properties

Compound	CAS REG #	MP	BP	VP	Sol.-H2O	Log P*
Rf-C(=O)F	335-64-8		131 °C			
Rf-CO2H	335-67-1	55 °C	189 °C	10 mm Hg	3.4 g/L	
Rf-CO2-NH4+	3825-26-1	130 °C (sub)	sublimes	1 x 10E-5 mm Hg	20 g/L gels	
Rf-C(=O)OMe	376-27-2		159 °C			
pH (1 g free acid /L Water) = 1.5 – 2.5			Free acid pKa is approximately 0.6			
Sodium or Silver salts of PFOA decompose above 250 °C to generate perfluoroolefins.						

- Surfactants traditionally emulsify octanol and water

2.0 Production of PFOA and its Salts

PFOA has been commercially manufactured by two major alternative processes: 1) the Simons Electro-Chemical Fluorination (ECF) process or 2) the telomerization process. The 3M Company was reported to be largest manufacturer and importer of PFOA and its salts in the United States in 1999. They predominantly used the ECF process to produce a PFOA precursor which is ultimately converted to PFOA. Data for the FC-26 (97% C8) suggests this purified material was about 80% linear.

In the ECF process to make PFOA, an electric current is passed through a solution of anhydrous hydrogen fluoride and an organic feedstock, typically an octanoic acid derivative. The ECF process replaces the carbon-hydrogen bonds on molecules of the organic feedstock with carbon-fluorine bonds, in an identical manner used to make PFOS. Perfluorination occurs when all the carbon-hydrogen bonds are replaced with carbon-fluorine bonds. The ECF process yields between 30-45 percent straight chain (normal) perfluorooctanoyl fluoride (PFOF), along with a variable mixture of byproducts and impurities. The product from the ECF process is not a pure chemical, but instead a mixture of isomers and homologs including higher and lower straight-chain homologs; branched-chain perfluoroalkyl fluorides of various chain lengths; straight-chain, branched, and cyclic perfluroalkanes and ethers; and other byproducts (3M Company, 2000a). After separation and recovery of desired material from the byproducts and impurities of the crude reaction mixture, the initially formed acid fluoride is base hydrolyzed in batch reactors and acidified to ultimately yield PFOA. The PFOA salts are synthesized by base neutralization of the acid to the salt in a separate reactor (3M Company, 2000b).

3M has characterized its manufacture of PFOA and its derivatives in 1997 at less than 500,000 kg per year in the US, and its importation at less than 100,000 kg (3M Company, 2000a). (These figures may overstate the total production volume of PFOA since the vast majority of PFOA is consumed in the manufacture of the ammonium or other metallic salts.) Industry sources have characterized 3M as the dominant global producer of PFOA-related chemicals, manufacturing approximately 85 percent or more of total worldwide volumes of the ammonium salt of PFOA

(FMG, 2001). USEPA has not located information that would contradict this claim.
More precise production volumes of PFOA and the ammonium and sodium salts have been reported to USEPA by 3M, but have been claimed as TSCA confidential business information, preventing disclosure in this report. Since 1985, USEPA has received a total of approximately 25 notifications for PFOA-related chemicals that were not previously on the TSCA Chemical Inventory. Most of these notifications were from companies other than 3M. In most cases, the notifications qualified for the Low Volume Exemption for new chemicals with a production volume less than 10 metric tons per year.

Current production volume information for manufacturers other than 3M has not been provided by industry, nor is it available in USEPA's Chemical Update System (which contains information on non-polymeric organic chemicals manufactured in the United States or imported in volumes above 4,525 kg). Furthermore, there is no information on the total cumulative production volumes of PFOA since initial commercialization. In terms of on-going production, in comments to the draft hazard assessment of PFOA 3M has stated that their May 16, 2000 announcement of the phase-out of the production of perfluorooctanyl chemistry and related products includes PFOA and its salts. This commits 3M to a complete phase-out of PFOA and PFOA-related chemicals identical to the phase out of PFOS and PFOS-related chemicals by the end of 2002. In addition, the Fluoropolymer Manufacturers Group (3M largest customer for PFOA products, specifically the ammonium salt, APFO) has informed the EPA that it will be manufacturing 300,000 kg/yr to meet the industry demand for APFO that 3M has not met (FMG 2002 AR226-1094). The FMG manufacturing process is based on telomerization.

The telomerization process, according to 3M's Bultman and Pike, " is the reaction of a telogen (such as pentafluoroethyl iodide...) with a polymerizable ethylenic compound (such as tetrafluoroethylene) to form 'telomers'. In this process the resultant telomer iodides are then reacted with ethylene, via free radical addition. This forms a mixture of iodides that can be reacted subsequently to form a variety of useful materials....". Dupont uses an analogous telomerization process. Either process yields predominantly pure straight-chain acids with an even number of carbon atoms. Commercial products manufactured through the telomerization process are generally mixtures of perfluorinated compounds with ranges of even carbon numbers (Renner, 2001). Distillation can be used to obtain pure components (ECT, 1994).

Aside from the United States, OECD Member countries that reportedly have production capacity include France, Germany, Italy, and Japan. There may also be some production in non-OECD countries such as China. Following are companies that may manufacture PFOA and its salts (3M Company, 2000b; Directory of World Chemical Producers, 1998; Dynax, 2000; Renner, 2001; SEMI, 2001):

<u>OECD</u>

- 3M Company (United States)
- DuPont (United States)
- Exfluor Research Corporation (United States)

- PCR Inc. (United States)
- Ciba Specialty Chemicals (Germany)
- Clariant (Germany)
- Dyneon (Germany)
- Hoechst Aktiengesellschaft (Germany)
- EniChem Synthesis S.p.A. (Italy)
- Miteni S.p.A. (Italy)
- Asahi Glass (Japan)
- Daikin (Japan)
- Dainippon (Japan)
- Tohkem Products Corporation (Japan)

Non-OECD

- Chenguang Research Institute of the Chemical Industry (China)
- Shanhai 3F New Materials Co., Ltd. (China)

2.1 Uses of PFOA and its Salts

PFOA is used mainly as a chemical intermediate, and its salts are used in emulsifier and surfactant applications.

According to 3M, the vast majority of PFOA is consumed to make the ammonium or sodium salts. 3M also uses PFOA as a reactive intermediate in the industrial synthesis of a fluoroacrylic ester. The fluoroacrylic ester is used in an industrial coating application (3M Company, 2000a).

The salts of PFOA have additional uses, mostly in surfactant and emulsifier applications. These include the following:

Processing aid in the industrial synthesis of fluoropolymers and fluoroelastomers such as polytetrafluoroethylene and polyvinylidene fluoride with a variety of industrial and consumer uses (3M Company, 2000a; DuPont, 2000; Daikin, 2001).

Post-polymerization processing aids in the stabilization of suspensions of fluoropolymers and fluoroelastomers prior to further industrial processing (3M Company, 2000a).

Processing aid for factory-applied fluoropolymer coatings on fabrics, metal surfaces, and fabricated or molded parts (3M Company, 2000a).

Extraction agent in ion-pair reversed-phased liquid chromatography (Petritis, 1999).

Based on the physicochemical properties of the salts of PFOA, they may also have other related surfactant or emulsifier uses as a photographic chemical or in the manufacture of electronic components such as semiconductors. These same properties may lead industry to explore PFOA

as a replacement chemical for PFOS in other applications in which PFOA is not currently used.

2.2 Environmental Fate

2.2.1 Photolysis

Direct photolysis of APFO was examined in two separate studies (Todd, 1979; Hatfield, 2001) and photodegradation was not observed in either study. In the Todd (1979) study, a solution of 50 mg/l APFO in 2.8 liters of distilled water was exposed to simulated sunlight at 22±2 ºC. Spectral energy was characterized from 290-600 nm with a max output at ~360 nm. Direct photolysis of the test substance was not detected. However, the author noted that sample purity was not properly characterized which may have contributed to experimental error.

In the Hatfield (2001) study, both direct and indirect photolysis were examined utilizing techniques based on EPA and OECD guidance documents. To determine the potential for direct photolysis, APFO was dissolved in pH 7 buffered water and exposed to simulated sunlight (Scrano, 1999; Nubbe, 1995). For indirect photolysis, APFO was dissolved in 3 separate matrices and exposed to simulated sunlight for periods of time from 69.5 to 164 hours. These exposures tested how each matrix would affect the photodegradation of APFO. One matrix was a pH 7 buffered aqueous solution containing H2O2 as a well-characterized source of OH radicals (Ogata, 1983; Lunak, 1992). This tested the propensity of APFO to undergo indirect photolysis. The second matrix contained Fe2O3 in water that has been shown to generate hydroxyl radicals via a Fenton-type reaction in the presence of natural and artificial sunlight (Kachanova, 1973; Behar, 1966). The third matrix contained a standard solution of humic material. Neither direct nor indirect photolysis of APFO was observed based on loss of starting material. Predicted degradation products were not detected above their limits of quantitation. There was no conclusive evidence of direct or indirect photolysis whose rates of degradation are highly dependent on the experimental conditions. Using the iron oxide (Fe2O3) photoinitiator matrix model, the APFO half-life was estimated to be greater than 349 days.

2.2.2 Volatility

Impinger studies were performed to examine the volatility of APFO and PFOS. Solutions of APFO or PFOS containing ammonium acetate in water/1-propanol (50:50) or phase transfer agents, e.g., n-alkyldimethylbenzylammonium chloride (3M Environmental Laboratory, 1993) were blown with 280 liters of air at a flow rate of 1 L/min. (3M Environmental Laboratory, 1993). The results indicate there is some loss of APFO and PFOS, but most of the solutions retained over 80% or more of the fluorochemicals. The average retention was 92% for both APFO and PFOS. This indicates that there is loss from the solutions. However, some of the solutions, particularly the n-alkyldimethylbenzylammonium chloride solution, appear to retain all the fluorochemicals. These results were reviewed by Dr. Edwin Tucker of the Chemistry Dept. at the University of Oklahoma (3M Environmental Laboratory, 1993). He concluded that it is very unlikely that these fluorochemicals were removed by bubbling air through water due to

their very low vapor pressures. He suggested that a more plausible mechanism for loss from the solution phase is concentration of the surfactants in foam and loss from the bubbled solutions as foam or micro-droplets.

In the second part of the experiment, air was passed over the fluorochemicals and bubbled through a train of impingers containing the ammonium acetate solution. It was expected that if any fluorochemicals were present in the air they would be transferred and retained by the ammonium acetate solution. However, no fluorochemicals were present in either the first or second impinger. The report concludes that the vapor pressure of both compounds is less than 10 E-07 mm Hg.

According to these experiments, APFO and PFOS (potassium salt) have very low volatility and vapor pressure. Quantitative conclusions regarding rates of volatilization from water or Henry's Law constant are not possible. However, APFO and PFOS are capable of transport out of water. Also, the loss of the fluorochemicals may have been as the free acids, not the salt forms. APFO sublimes at 130 C (see Physicochemical Properties Section 1.1). There is no information on the validity of the test method for determining volatility of the test substance. The study also lacks characterization of the purity of the test substance.

2.2.3 Biodegradation

Using an acclimated sludge inoculum, the biodegradation of APFO was investigated using a shake culture study modeled after the Soap and Detergent Association's presumptive test for degradation (Reiner, 1978). Both thin-layer and liquid chromatography did not detect the presence of any metabolic products over the course of 2 1/2 months indicating that PFOA does not readily undergo biodegradation. In a related study, 2.645 mg/L APFO was not measurably degraded in activated sludge inoculum (Pace Analytical, 2001). Test flasks were prepared using a mineral salts medium, 1 mL methanol, and 50 mL settled sludge. Analysis was conducted with a HPLC/MSD system. Several other studies conducted between 1977-1987 also did not observe APFO biodegradation using what probably were standard COD and BOD methods, however, the methods used in these studies were either insufficiently described (i.e. no description of experimental protocols) or there were indications of a high degree of experimental error. The results were, therefore, deemed unreliable by the submitter (3M Company, 1977; 3M Company, 1980; 3M Company, 1985b; Pace Analytical, 1997).

2.2.4 Hydrolysis

The 3M Environmental Laboratory (2001a) performed a study of the hydrolysis of PFOA. The study procedures were based on EPA's OPPTS Guideline Document 835.2110 (EPA 1998); although the procedures do not fulfill all the requirements of the guideline, they were more than adequate for these studies. Results were based on the observed concentrations of PFOA in buffered aqueous solutions as a function of time. The chosen analytical technique was high performance liquid chromatography with mass spectrometry detection (HPLC/MS).

During the study, samples were prepared and examined at six different pH levels from 1.5 to 11.0 over a period of 109 days. Experiments were performed at 50°C and the results extrapolated to 25°C. Data from two of the pH levels (3.0 and 11) failed to meet the data quality objective and were rejected. Also rejected were the data obtained for pH 1.5 because ion pairing led to artificially low concentrations for all the incubation periods. The results for the remaining pH levels (5.0, 7.0, and 9.0) indicated no clear dependence of the degradation rate of PFOA on pH. From the data pooled over the three pH levels, it was estimated that the hydrolytic half-life of PFOA at 25°C is greater than 92 years, with the most likely value of 235 years. From the mean value and precision of PFOA concentrations, it was estimated the hydrolytic half-life of PFOA to be greater than 97 years.

2.2.5 Bioaccumulation

Three studies have been conducted to determine the potential for bioaccumulation of APFO. Howell et al. (1995) exposed Fathead minnows to 25 mg/l APFO for 13 days. After 13 days exposure, the fish were then removed from APFO contaminated water and analyzed for depuration over 15 days. After 192 and 312 hours exposure to APFO contaminated water, the average concentration of APFO in fish tissue was 44.7 and 46.7 μg/g wet weight (ww), respectively. At this point, APFO appeared to reach steady state. Twenty-four hours after being transferred to clean water, the concentration of APFO decreased to 19.9 μg/g ww and by 96 hours post-exposure, the concentration had decreased to approximately 8 μg/g ww and remained relatively constant until test termination at 360 hours. The calculated BCF for APFO was 1.8. It should be noted that questions have been raised about this study regarding the analytical techniques, high-test chemical concentration, and short test duration.

In a bioaccumulation study of carp (*Cyprinus carpio*), 28 carp were exposed to two concentrations of PFOA, 5 and 50 : g /L respectively for 28 days in a flow through system (Kurume Laboratory, 2001). The chemical was prepared for exposure by using a dispersant made up of hydrogenated castor oil and mixed in acetone. The purity of PFOA was 100%. Water quality was monitored daily throughout the duration. The bioaccumulation potential of PFOA was found to be low. The carp exposed to 5 : g/L resulted in a BCF of 3.1 while the carp exposed to 50 : g /L showed a BCF of < 5.1 – 9.1.

Vraspir (1979) conducted a study to determine if bluegill sunfish bioaccumulate fluorochemicals from the 3M Decatur plant. Two lots of 30 fish were used. One lot was exposed to Decatur plant effluent for 21 days and the other to river water only for 23 days. Exposed fish, both living and dead, as well as the control fish were homogenized and analyzed for fluorochemicals by GC, TLC, and GC/MS. There were no detectable amounts of APFO in the ethyl acetate or toluene extracts of the tissues. No fluorochemicals were detected in the river water exposed fish. However, interpretation of this study is problematic for several reasons. Effluent concentrations of subject fluorochemicals were not characterized and the specific protocol for exposure of the fish was not found. There was also no information on analysis of the Tennessee River water or effluent used in the study. Additionally, it was not known if there was any opportunity for the depuration of the fish prior to sacrifice. No explanation was attempted as to what was the cause

of the twelve dead fish in the effluent-exposed group. The study also did not differentiate between the bioaccumulation of the test compound and the sorption onto the surface of the fish.

2.2.6 Soil Adsorption

The adsorption-desorption of APFO was studied in 25 ml solutions of 14C-labeled APFO in distilled water with 5 g Brill sandy loam soil for 24 hours at a temperature of 16-19 ºC. The study reported a Kd of 0.21 and a Koc of 14 indicating that PFOA has high mobility in Brill sandy loam soil (Welsh 1978). The Koc value, however, is questionable due to the lack of accurate information on the purity of the 14C-labeled test substance (Boyd 1993a,b).

Moody and Field (1999) conducted sampling and analysis of samples taken from groundwater 1 to 3 meters below the soil surface in close proximity to two fire-training areas with a history of aqueous film forming foam use. Perfluorooctanoate was detected at maximum concentrations ranging from 116 to 6750 ug/L at the two sites many years after its use at those sites had been discontinued. These results suggest that APFO may have the potential to migrate through soils to relatively shallow groundwater where it persists.

2.3 Environmental Exposure

2.3.1 Discharge to Air

For 1997, 3M estimated 1950 lbs. of PFOA-compound (PFOA and related salts) stack releases at its Cottage Grove, MN location and another 4500 lbs. from Cottage Grove incinerated offsite (3M, 2000). In 1998, 70% of the fluoride-containing wastes at 3M's Decatur location were incinerated off-site; incineration is now the primary disposal method for these materials (3M, 2000). For 1999, DuPont estimated stack releases of 24,000 lbs. APFO at its Washington Works, WV location, plus another 16,000 lbs. from Washington Works incinerated offsite (DuPont, 2000).

The above data may not account for all of the sources of air releases of PFOA. Other potential sources addressed in the literature include thermolysis of fluoropolymers (Ellis et al, 2001); the low temperature of sublimation (~135C) that would yield PFOA from drying, aka sintering, of fluoropolymer made with PFOA as a processing aid; the thermally refractive character of the C-F bond (Napoli, et al, 1984; Taylor, et al, 1990; Tsang, et al, 1998) that would require very high temperature incineration as a means of destruction of this molecule.

2.3.2 Discharge to Water

By analogy to PFOS, PFOA discharged to water may remain there, become adsorbed to particulate matter and sediment, and/or be assimilated by organisms. For 1999, 3M estimated PFOA-compound water releases of <30,000 lbs. at its Decatur AL location, and <15,000 lbs. at its Cottage Grove MN location (3M Company, 2000a,b). For 1999, DuPont estimated the following APFO water releases per location: Washington Works WV, 55,000 lbs; Parlin NJ, 300

lbs.; Spruance VA, 150 lbs.; Chambers Works NJ, 9500 lbs. (DuPont, 2000).

DuPont measured an APFO concentration at its site Washington Works, WV site of 0.552 ug/l from a 1999 drinking water sample obtained from GE Plastics immediately downstream on the Ohio River. Modeled 1996 APFO-compound releases indicated an average annual PFOA concentration of 0.423 ug/l, with APFO concentrations likely to exceed 1 ug C-8/l about 50% of the time during the year, and likely to exceed 10 ug APFO/L about 2.2% of the time during the year.

2.3.3 Discharge to Land

3M reported that land treatment of sludge from wastewater treatment at their Decatur, AL location ended in mid-1998; less than 500 lbs. were disposed to land at that site in 1997. Sludge from the Decatur site is now transported to an offsite landfill; sludge from 3M's Cottage Grove, MN facility is sent to an industrial landfill (3M Company, 2000a,b). DuPont (2000) estimated 3,900 lbs. of APFO sludge landfilled on site in 1999 at their Chambers Works, NJ facility. DuPont estimated 2,600 lbs. APFO transferred offsite to a hazardous waste landfill from their Washington Works, WV facility.

Prior operations resulted in ground- and surface water concentrations of APFO monitored at three landfills operated by DuPont's Washington Works WV facility. Average surface water concentrations for two landfills were 1392 ug/L and 18.5 ug/L, respectively. A third landfill had a maximum concentration of 33 ug/L in the permitted outfall. Average groundwater concentrations for two landfills were 2537 ug/L and 8.83 ug/L, respectively. A third landfill had a maximum groundwater concentration of 15 ug/L (DuPont, 2000).

DuPont also reported the following APFO concentrations, measured January 2000, in three drinking water wells of the Lubeck Public Service District, downstream of DuPont's Washington Works WV site: 0.8 ug/L, 0.44 ug/L, and 0.313 ug/L (DuPont, 2000). As of August 2000, the Lubeck Public Service District (LPSD) reported APFO concentrations of 0.2 ppb in drinking water at DuPont's Washington Works facility, and 0.2, 0.5, and 0.1 ppb in the three LPSD wells (LPSD, 2000).

2.3.4 Environmental Monitoring

3M's Multi-City Study reported on PFOA concentrations from water, sludge, sediment, POTW effluent, and landfill leachate samples taken in six cities (3M, 2001a). Four of the cities (Decatur, AL; Mobile, AL; Columbus, GA; Pensacola, FL) were "supply" cities that have manufacturing or industrial use of fluorochemicals; two of the cities (Cleveland, TN; Port St. Lucie, FL) were "control" cities that do not have significant fluorochemical activities. Across all cities, POTW effluent concentrations ranged from 0.040 to 2.42 ppb. The POTW sludge (dry wt.) range was non-detect to 244 ppb; the drinking water range was non-detect to 0.029 ppb; the landfill leachate range was non-detect to 48.1 ppb; the surface water range was non-detect to 0.083; the sediment range was non-detect to 1.75 ppb (dry wt.); and the quiet water range was

non-detect to 0.097 ppb. The "control" cities samples generally inhabited the lower end of the above ranges, except for the POTW effluent and sludge findings for Cleveland, which were intermediate in their ranges.

The Multi-City Study also included a market basket sampling of PFOA residue in a total of over 200 samples taken from green beans, apples, pork muscle, cow milk, chicken muscle, chicken eggs, bread, hot dogs, catfish, and ground beef (3M, 2001a). Measurable quantities of PFOA, ranging up to 2.35 ng/g, were found in two ground beef samples (control cities), two bread samples (control and supply cities), two apple samples (supply cities), and one green bean sample (supply city).

Giesy reported that PFOA was rarely found in fish and fish-eating water birds. Fish were sampled from the U.S., certain European countries, the North Pacific Ocean, and Antarctic locations (Giesy, 2001a). Fish-eating bird samples were collected from the U.S., including Midway atoll, the Baltic and Mediterranean Seas, Japanese and Korean coasts (Giesy, 2001b)

Giesy reported on PFOA in mink and river otter livers from the U.S. (Giesy, 2001c). PFOA was found in a few mink livers from Massachusetts at a concentration range of <18 to 108 ng/g, dry wt., but not found in mink from Louisiana, South Carolina and Illinois. PFOA concentrations in river otter livers from Washington and Oregon States were less than the quantification limit of 36 ng/g, wet wt.

Giesy reported that PFOA was not detected at quantifiable concentrations in oysters collected in the Chesapeake Bay and Gulf of Mexico of the U.S. coast (Giesy, 2001d).

Giesy reported on the concentrations of PFOA in surface water, sediments, clams, and fish collected from locations upstream and downstream of the 3M facility at Decatur, AL (Giesy, 2001e). Of the three downstream water and sediment sampling locations, the two closest to the 3M facility had PFOA surface water concentrations significantly greater than the two upstream sites (means of 1900ug/L and 1024 ug/L, vs. 0.008 (est.) and 0.028 ug/L); the three downstream locations also had sediment concentrations significantly greater than the upstream sites (wet wt. means 1855 ug/kg, 892 ug/kg, 238 ug/kg vs. 0.08(est.) and 0.09(est.)). The average fish whole body PFOA concentration for the upstream location was 11.7 ug/kg (wet wt.), while that for the downstream location was 106.4 ug/kg; the small sample size prevented determination of significance. The average PFOA concentration in clams at the upstream location was 4.38 ug/kg; that for the downstream location was 8.42 ug/kg. These differences were not significant.

Hansen (2002) reported concentrations of PFOA measured from surface water samples taken from the Tennessee River up- and downstream of the outfall from the fluorochemical manufacturing facility at Decatur AL (the 3M facility mentioned above). There were 20 sampling sites above and 20 sites below the outfall location, spaced at approximately 2 mile intervals. None of the samples taken from upstream of the facility had measurable concentrations of PFOA. The downstream concentrations of PFOA were observed to increase at a point approximately six miles below the outfall; the average PFOA concentration from that

point downstream was 394 ± 128 ng/L. The report states that the consistency of the PFOA concentrations within these two regions suggests the absence of either major environmental sinks or additional sources of PFOA in the areas sampled.

Moody reported the concentrations of PFOA in surface water from stream locations sampled upstream and downstream of a spill of fire-fighting foam that contained perfluorinated surfactants, including PFOA (Moody et al., 2002). Upstream surface water samples taken over the three week period post-spill had PFOA levels of 0.008-0.033 ug/L; corresponding downstream levels ranged from 0.035-10.6 ug/L. Statistical significance was not evaluated for these measurements.

2.4 Human Biomonitoring

Table 2 provides serum PFOA levels in both occupational cohorts and in the general population. The highest levels reported to date in the general population are similar to some of the lowest levels in workers exposed to PFOA occupationally. The data are currently limited to those discussed below.

3M has been offering voluntary medical surveillance to workers at plants that produce or use perfluorinated compounds since 1976. Serum PFOA levels have been measured and reported since 1993. Prior to this time, only total organic fluorine was measured. The results of biomonitoring for PFOA have been reported for 3 plants: Cottage Grove, MN; Decatur, AL and Antwerp, Belgium. Surveillance years include 1993, 1995, 1997, 1998, and 2000, although not all of the plants offered surveillance in all of these years. The 1998 data reported for the Decatur plant consist of a random sample of employees; however, volunteers participated in all of the other sampling periods for all of the plants.

Mean serum PFOA levels have increased slightly at both the Cottage Grove and Decatur plants since 1993. Workers at the Cottage Grove plant, where PFOA exposures are highest, have the highest PFOA serum levels. The latest sample was in 1997 (Olsen, et al., 1998b). The mean serum PFOA level was 6.4 ppm (range = 0.1 – 81.3 ppm). Only 74 employees participated in the 1997 surveillance. The eligible voluntary participation rates ranged from approximately 50% in 1997 to 70% in 1993.

At the Decatur plant, 263 of 500 employees participated in 2000 (Olsen, et al., 2001a). The mean serum PFOA level was 1.78 ppm. It was higher in males (n = 215) than females (n = 48), 1.90 and 1.23 ppm, respectively. In addition, male production employees had higher mean serum levels (2.34 ppm). Five employees had serum levels greater than 5 ppm, the Biological Limit Value established by the 3M Exposure Guideline Committee. Cell operators had the largest increase in serum PFOA between 1998 and 2000. The highest level was in a chemical operator on the Scotchgard team (12.70 ppm). The mean level for the rest of the members of the team was 5.06 ppm (range 5 - 9 ppm). Other job categories did not exhibit such a large increase. 3M reports that this is due to increased PFOA production at the Decatur plant beginning in 1999.

Serum PFOA levels for the Antwerp plant are lower than at Decatur or Cottage Grove, and have decreased slightly since 1995 (Olsen, et al., 2001b). Participation in medical surveillance at the Antwerp plant was the highest it had ever been in 2000 (258 volunteers out of 340 workers). The mean serum PFOA level was 0.84 ppm, and the highest serum level reported was 7.04 ppm. As in the Decatur plant, males (n = 209) had higher mean serum PFOA levels (1.03 ppm) than females (n = 49, 0.07 ppm). Three employees had levels greater than 5 ppm.

3M's Specialty Materials Manufacturing Division Laboratories, where employees perform fluorochemical research (Building 236), conducted voluntary biomonitoring of 45 employees in 2000 (Olsen, et al., 2001c). The mean PFOA serum level was 0.106 ppm (range 0.008 – 0.668 ppm).

Serum PFOA levels in corporate staff and managers at a 3M plant in St. Paul, MN, where occupational exposure to PFOA should not have occurred, were reported (3M Report, 1999). Four of 31 employees had serum PFOA levels greater than the detection limit of 10 ppb. The mean for these employees was 12.5 ppb.

Data on PFOA levels in the general population are very limited. They are very recent so that trends over time cannot be established. The mean serum PFOA levels are lower in the general population than in workers exposed to PFOA.

Pooled blood samples from U.S. blood banks indicate mean PFOA levels of 3 to 17 ppb (3M Company, Feb. 5, 1999; 3M Company, May 26, 1999). The highest pooled sample reported was 22 ppb. Samples were collected in 1998 and 1999. However, it cannot be assumed that these levels are generalizable to the U.S. population for several reasons: 1) blood donors are a unique group that does not necessarily reflect the U.S. population as a whole, 2) many of the blood banks originally contacted for possible inclusion in the study declined to participate, 3) only a small number of samples have actually been analyzed for PFOA, and 4) no other data such as age, sex, or other demographic information are available on the donors.

Individual blood samples from 3 different age populations were recently analyzed for PFOA and other fluorochemicals using high-pressure liquid chromatography/electrospray tandem mass spectrometry (HPLC/ESMSMS) (Olsen, et al., 2002a, 2002b, 2002c). The studies' participants included adult blood donors, an elderly population participating in a prospective study in Seattle, WA, and children from 23 states participating in a clinical trial. Overall, the PFOA geometric means were similar across all 3 populations (4.6 ppb, 4.2 ppb, and 4.9 ppb, respectively). The geometric means and 95% tolerance limits (the proportion of the population expected to be found) and their upper bounds were comparable across all 3 studies. However, the upper ranges for the children and adults were much higher than for the elderly population. It is not clear whether this is the result of geographic differences in PFOA levels or some other factor. It should be noted that PFOS and PFOA were highly correlated in all three studies ($r = .63$, $r = .70$, and $r = .75$) and that PFOA did not meet the criteria for a log normal distribution based on the Shapiro-Wilk test in any of the studies. The authors suggest that it may be due to the greater

proportion of subjects with values < LLOQ; however, only geometric means were reported. The details of each study are provided below.

Blood samples from 645 U.S. adult blood donors (332 males, 313 females), ages 20-69, were obtained from six American Red Cross blood banks located in: Los Angeles, CA; Minneapolis/St. Paul, MN; Charlotte, NC; Boston, MA; Portland, OR, and Hagerstown, MD (Olsen, et al., 2002a). Each blood bank was requested to provide approximately 10 samples per 10-year age intervals (20-29, 30-39, etc.) for each sex. The only demographic factors known for each donor were age, gender, and location.

The geometric mean serum PFOA level was 4.6 ppb. The range was <lower limit of quantitation (1.9) to 52.3 ppb. Males had significantly higher ($p < .05$) geometric mean PFOA levels than females (37.8 ppb vs. 32.1 ppb). Age was not an important predictor of adult serum fluorochemical concentrations. When stratified by geographic location, the highest geometric mean for PFOA was in the samples from Charlotte, NC (6.3 ppb, range: 2.1 – 29.0) and the lowest from Portland (3.6 ppb, range: 2.1 – 16.7). The highest individual value was reported in Hagerstown (52.3 ppb).

Serum PFOA levels were reported for 238 (118 males and 120 females) elderly volunteers in Seattle participating in a study designed to examine cognitive function in adults aged 65-96 (Olsen, et al., 2002b). Age, gender and number of years' residence in Seattle were the only data available on the participants. Most of the participants were under the age of 85 and had lived in the Seattle area for over 50 years.

The geometric mean of PFOA for all samples was 4.2 ppb (95% CI, 3.9 – 4.5). The range was 1.4 – 16.7 ppb. There was no significant ($p < .05$) difference in geometric means for males and females. In simple linear regression analyses, age was negatively ($p < .05$) associated with PFOA in elderly men and women. In bootstrap analyses, the mean of the 95% tolerance limit for PFOA was 9.7 ppb with an upper 95% confidence limit of 11.3 ppb. PFOS and PFOA were highly correlated ($r = .75$) in this study.

A sample of 598 children, ages 2-12 years old, participating in a study of group A streptococcal infections, was analyzed for serum PFOA levels (Olsen,et al., 2002c). The samples were collected in 1994-1995 from children residing in 23 states and the District of Columbia. PFOA did not meet the criteria for a log normal distribution based on the Shapiro-Wilk test. The authors suggest that it may be due to the greater proportion of subjects with values < LLOQ for PFOA. However, only geometric means were reported. The geometric mean of PFOA for all of the participants was 4.9 ppb (95% CI, 4.7 – 5.1). The range was 1.9 to 56.1 ppb. Male children had significantly ($p<.01$) higher geometric mean serum PFOA levels than females: 5.2 ppb and 4.7 ppb, respectively. In simple linear regression analyses, age was significantly ($p < .05$) negatively associated with PFOA in both males and females. When stratified by age, the geometric mean of PFOA was highest at age 4 (5.7 ppb) and lowest at age 12 (3.5 ppb). Although the data were not reported, a graphical presentation of log PFOA levels for each state by gender looked similar across the states; however, it is difficult to interpret these data without

the data and given the limited sample size for each gender/location subgroup. In bootstrap analyses, the mean of the 95% tolerance limit for PFOA was 10.1 ppb with an upper 95% confidence limit of 11.0 ppb. PFOS and PFOA were highly correlated (r = .70) in this study. PFOA and PFHS (perfluorohexanesulfonate) were also correlated (r = .48).

The above 3 studies indicate similar geometric means and ranges of PFOA among sampled adults, children, and an elderly population. However, an unexpected finding was the level of PFHS and M570 (N-methyl perfluorooctanesulfonamidoacetate) in children. These serum levels were much higher in the sampled children than in the sampled adults or elderly. It is not clear why this occurred, but it is probably due to a different exposure pattern in children.

In another study, the PFOA concentration was analyzed in human sera and liver samples (Olsen et al., 2001d). Thirty-one donor samples were obtained from 16 males and 15 females over an 18-month period from the International Institute for the Advancement of Medicine (IIAM). The average age of the male donors was 50 years (SD 15.6, range 5-69) and the average age of the female donors was 45 years (SD 18.5, range 13-74). The causes of death were intracranial hemorrhage (n = 16 or 52%), motor vehicle accident (n = 7 or 23%), head trauma (n = 4 or 13%), brain tumor (n = 2 or 6%), drug overdose (n = 1 or 3%) and respiratory arrest (n = 1 or 3%). Both serum and liver tissue were obtained from 23 donors; 7 donors contributed liver tissue only and 1 donor contributed serum only. Serum samples were obtained from 5 ml of blood; liver samples consisted of 10 g of tissue. Samples were frozen at IIAM and shipped frozen to 3M for analysis. Samples were extracted using an ion-pairing extraction procedure and were quantitatively assayed using HPLC-ESMSMS and evaluated versus an unextracted curve. Extensive matrix spike studies were performed to evaluate the precision and accuracy of the extraction procedure. Serum values for PFOA ranged from < LOQ (<3.0) – 7.0 ng/mL. Assuming the midpoint value between zero and LOQ serum value for samples <LOQ, the mean serum PFOA level was 3.1 ng/mL with a geometric mean of 2.5 ng/mL. No liver to serum ratios were provided because more than 90% of the individual liver samples were <LOQ.

Table 2. Serum PFOA Levels in Human Populations

Occupational Exposures – Serum Levels (ppm)				
Plant	**Arithmetic Mean**	**Range**	**Geometric Mean**	**95% CI**
Cottage Grove Plant				
1997 (n = 74)	6.4	0.1 – 81.3	*	*
1995 (n = 80)	6.8	0.0 – 114.1	*	*
1993 (n = 111)	5.0	0.0 – 80.0	*	*
Decatur Plant				
2000 (n = 263)	1.78	0.04 – 12.70	1.13	0.99 – 1.30
1998 (n = 126)	1.54	0.02 – 6.76	0.90	0.72 – 1.12
1997 (n = 84)	1.57	not reported	*	*
1995 (n = 90)	1.46	not reported	*	*
Antwerp Plant				
2000 (n = 258)	0.84	0.01 – 7.04	0.33	0.27 – 0.40
1995 (n = 93)	1.13	0.00 – 13.2	*	*
Building 236				
2000 (n = 45)	0.106	0.008 – 0.668	0.053	0.037 – 0.076

General Population Exposures – Serum Levels (ppb)				
Source	**Arithmetic Mean**	**Range**	**Geometric Mean**	**95% CI**
Pooled samples				
Commercial sources of blood, 1999 (n = 35 lots)	3	1 - 13	*	*
Blood Banks (n = 18), 1998 ~340-680 donors	17**	12 - 22	*	*
Individual samples				
American Red Cross blood banks, 2000 (n = 645)	5.6	1.9 - 52.3	4.6	4.3 – 4.8
Elderly (ages 65-96), 2000 (n = 238)	not reported	1.4 – 16.7	4.2	3.9 – 4.5
Children (ages 2-12), 1995 (n= 598)	5.6	1.9 - 56.1	4.9	4.7 – 5.1
3M Corporate managers/staff St. Paul, MN, 1998 (n = 31)	12.5***	not reported	*	*

*Geometric mean and 95% confidence intervals were not included in the reports.
**PFOA detected in about 1/3 of the pooled samples but quantifiable in only 2
***only 4 employees were above the detection limit of 10 ppb

3.0 Human Health Hazards

3.1. Metabolism and Pharmacokinetics

3.1.1 Half-life in Humans

There are very limited data on the half-life of PFOA. With the exception of a 1980 study in which total organic fluorine in blood serum was measured in one worker, no other data were available until June 2000 (Ubel et al., 1980). A half-life study on 27 retirees from the Decatur and Cottage Grove 3M plants was undertaken, in which serum samples were drawn every 6 months over a 5-year period. Two interim reports describing the results thus far have been submitted (Burris et al., 2000; Burris et al., 2002). The first interim report suggested a median serum half-life of PFOA of 344 days, with a range of 109 to 1308 days. The two highest half-life calculations were for the 2 female retirees who participated in this study (654 and 1308 days).

There were several limitations to this first analysis including: 1) the limited data available and the range of serum PFOA levels measured; 2) serum was analyzed after each collection period with only one measurement per time period on different days using slightly different analytical techniques; and 3) the reference material purity was not determined until after the first 3 samples had been analyzed. An effort was made to minimize experimental error, including systematic and random error in the analytical method, involving 9 of the original 27 subjects. Serum samples were collected from each of the subjects over 4 time periods spanning 180 days, measured in triplicate with all time points from each subject analyzed in the same analytical run. This would allow for statistical evaluation of the precision of the measurement and assure that all systematic error inherent in the assay equally affected each sample used for half-life determination.

Of the 9 retirees included in this analysis, there were 7 males and 2 females, all from the Decatur plant. The average age of the retirees was 61 years, the mean number of years worked at Decatur was 27.7 years, and the average number of months retired was 18.9. Average BMI of this group was 27.9. The mean PFOA value at study initiation was 0.72 ppm (range 0.06 – 1.84 ppm, SD = 0.64).

The mean serum half-life for PFOA was 4.37 years (range 1.50 – 13.49 years, SD = 3.53). Only 1 employee had a half-life value that exceeded 4.3 years. The 2 females had values of 3.1 and 3.9 years. Age, BMI, number of years worked or years since retirement were not significant predictors of serum half-lives in multivariable regression analyses.

This analysis has attempted to reduce experimental error in the determination of a half-life for PFOA. However, several issues should be noted. First, the effect of continued non-occupational, low-level exposure on the half-life is unknown. Second, because subjects' blood contained concentrations of fluorochemicals that varied by a factor of 30, the data cannot be pooled or averaged unless the serum concentration decay curve shows first-order kinetics.

Third, it is not known if there are interactions between PFOA and other fluorochemicals in the body. Fourth, this estimate is much higher than that reported in lab animals. Fifth, systematic error of the analytical method could be as high as +/- 20% and still satisfy the data quality criteria.

3.1.2 Absorption Studies in Animals

APFO is well absorbed following oral and inhalation exposure, and to a lesser extent following dermal exposure. The liver and serum levels have been measured in several subchronic and reproductive toxicity studies. These results are presented in the relevant studies summarized in subsequent sections. Other studies are summarized here.

In female rats, an average of 749 ug or 37% of the fluorine in the administered dose was recovered in the urine within 4.5 hr after PFOA dose (by stomach intubation 2 ml of an aqueous solution containing 2 mg PFOA) (Ophaug and Singer, 1980). The quantity of nonionic fluorine recovered in the urine increased to 61% of the dose at 8 hr, 76% at 24 hr, and 89% at 96 hr.

After a single oral dose of ^{14}C-PFOA (mean dose, 11.0 mg/kg) in solution to groups of three male rats, at least 93% of the total carbon-14 was absorbed at 24 hours (Gibson and Johnson, 1979). The half-life for elimination of total carbon-14 from plasma was 4.8 days.

Following APFO head-only inhalation exposure in male rats (6 hr/day, 5 days/wk for 2 wk to 0, 1, 8 or 84 mg/m3) concentrations of organofluoride in the blood showed a dose relationship with initial levels of 108 ppm in rats treated at 84 mg/m3 (Kennedy et al., 1986). Immediately after the tenth exposure period, the mean organofluoride blood levels were 13 ppm, 47 ppm, and 108 ppm in the 1, 8, and 84 mg/m3 dose groups.

Subchronic dermal APFO treatment in rats and rabbits (10 applications, 5 doses, 2 rest days, 5 doses) with either 0, 20, 200, or 2000 mg/kg resulted in elevated blood organofluorine levels which increased in a dose-related manner (Kennedy, 1985).

O'Malley and Ebbins (1981) conducted a range finding study which indicates significant dermal absorption of PFOA in male and female rabbits. PFOA (100 mg/kg, 1000 mg/kg, and 2000 mg/kg in saline slurry) was applied to approximately 40% of the shaved trunk of the animals, which were then fitted with a plastic collar, and the trunk was wrapped with impervious plastic sheeting. The exposure period was 24 hr, 5 days/week over 14 days. Mortality was 100% (4/4) in the 2000 mg/kg group, 75% (3/4) in the 1000 mg/kg group and 0% (0/4) in the 100 mg/kg group.

A tetrabutyl ammonium salt of perfluorooctanoate in the form of treated fabric and as a liquid formulation was applied dermally to rabbits (Johnson, 1995b). Liver samples were analyzed at 28 days post dose for total organic fluorine. The results from treated animals were the same as control values. All total organic values were below the practical quantitation limit. Serum levels were also below the practical quantitation limits of the analysis for samples

collected at day 1 and 2 after administration of the mixture or the treated fabric. From the pharmacokinetic study (Johnson, 1995a), it would be unlikely that any extent of absorption could have been detected in this study.

3.1.3 Distribution Studies in Animals

PFOA distributes primarily to the liver, plasma, and kidney, and to a lesser extent, other tissues of the body. It does not partition to the lipid fraction or adipose tissue, but does bind to macromolecules in the tissues. There is evidence of enterohepatic circulation of the compound.

Serum and liver concentrations of PFOA were determined in rhesus monkeys in a 90 day oral toxicity study (Griffith and Long, 1980). In monkeys at the 3 mg/kg/day dose, mean serum PFOA was 50 ppm in males and 58 ppm in females. At the same dose, males had 3 ppm and females 7 ppm in liver samples. At 10 mg/kg/day doses, male monkeys had a mean serum PFOA of 58 ppm and females 75 ppm. Liver levels were 9 and 10 ppm for males and females, respectively, measured as organic fluoride.

Ophaug and Singer (1980) measured ionic fluoride and total fluorine in the serum of female rats following the administration of PFOA by stomach intubation (2 ml of an aqueous solution containing 2 mg PFOA). Serum from rats 4.5 hr after the administration of PFOA had a nonionic fluorine level 13.6 ppm and virtually all of this was bound to components in the serum and not ultrafilterable. Despite the large increase in nonionic fluorine in the serum, the ionic fluoride level remained very low (0.03 ppm). Prior to intubation of PFOA, the ionic and nonionic fluorine levels in serum were 0.032 and 0.07 ppm, respectively. The nonionic fluorine level in the serum decreased to 11.2 ppm at 8 hr, 0.35 ppm at 24 hr, and 0.08 ppm at 96 hr. The authors conclude that PFOA is rapidly absorbed from the gastrointestinal tract and rapidly cleared from the serum.

Twenty-four hours after oral administration of APFO (2 mg APFO in 2 ml aqueous solution by stomach intubation), female rats had a mean serum nonionic fluorine level of 0.35 ppm, while male rats had a mean serum nonionic fluorine level of 44.0 ppm (Hanhijarvi et al., 1982). APFO was bound to a similar extent in the plasma of male and female rats (97.5% bound).

In male and female rats administered ^{14}C-PFOA in propylene glycol/water (9.4 umol/kg, i.p.), the concentration of ^{14}C-PFOA-derived radioactivity in the blood was higher and eliminated more slowly in males (t1/2=9 days, males vs 4 hr, females, Vanden Heuvel et al., 1991). In the male rats, the liver had the highest PFOA concentration (21% of dose at 2 hr, 2% of dose at 28 days) followed by the plasma and kidney. Far lower PFOA concentrations were found in the heart, testis, fat, and gastrocnemius muscle. In females at 2 hr post dose, the highest concentrations of PFOA were found in the plasma followed by the kidney, liver and ovaries in that order. The average t1/2 for elimination of PFOA from the liver in male rats was 11 days compared to an average of 9 days for extrahepatic tissues. In females, the average t1/2 for tissue elimination was approximately 3 hr.

Vanden Heuvel et al. (1991) investigated the disposition of PFOA in perfused male rat liver. Approximately 11% of the cumulative dose of ^{14}C-PFOA infused (0.08 umol/min x 48 min, 3.84 umol total) was extracted by the liver during a first pass. In addition, the cumulative percent of PFOA extracted by the liver at 2 min (33%) was substantially greater than that seen after 48 min (11%) indicating that first-pass hepatic uptake of PFOA may be saturable.

Ylinen et al. (1990) studied the difference between male and female Wistar rats in the distribution and accumulation of PFOA after a single and subchronic administration. The single dose of PFOA (50 mg/kg in propylene glycol-water mixture, 1:1, vol. 0.25 ml/100g) was administered intraperitoneally to 10 week old rats (20 male, 20 female). Subchronic administration of PFOA consisted of 3, 10, and 30 mg/kg/day by gavage (in 0.9% NaCl, 0.5 ml/100g) to newly weaned rats (18 male, 18 female). After the single dose, samples were collected for PFOA determination 12, 24-168 (at 24 hr intervals), 244 and 336 hours after the administration, and in the subchronic test on the 28th day. The serum was collected by cardiac puncture; after decapitation the brain and at necropsy samples from the liver, kidney, lung, spleen, ovary, testis, and adipose tissue were collected and frozen. The biological half-life of PFOA in the serum and tissues was determined from the linear relationship between time and PFOA concentration in the semilogarithmic plot. In the single-dose study, concentration of PFOA in the serum and tissues was higher in males than females at all time periods. Twelve hours after the administration of PFOA about 10% of the dose was found in the serum of females, whereas about 40% was in the serum of males. After 14 days about 3.5% of the dose remained in the serum. In females, PFOA concentration in the serum, liver, and kidney occurred in a discontinuous fashion, indicating distinct phases. The half-life in the serum was 24 and 105 h in the females and males, respectively. In the females, a half-life of 60 hr was estimated in the liver during the first week. In the males, the half-life in liver was 210 hr. Although PFOA was retained by the liver, it was not found in the lipid fraction. In the kidney, the half-life was 145 hr and 130 hr in females and males, respectively. In the spleen, the half-life was 73 hr and 170 hr in females and males, respectively. PFOA was also found in brain tissue. PFOA was not detectable in adipose tissue. In the subchronic study, samples taken on the 28th day indicated significantly higher PFOA concentrations in the serum and tissues of males versus females in all three dose levels. After subchronic, as well as single-dose administration, PFOA was mainly distributed in the serum of rats. High concentrations of PFOA were also found in the liver, kidney, and lung of males and females. At the high dose level (30 mg/kg/day), females and males exhibited, respectively, serum concentrations of 13.92 and 51.65 ug/ml, liver concentrations of 6.64 and 49.77 ug/g, kidney concentrations of 12.54 and 39.81 ug/g, spleen concentrations of 1.59 and 4.10 ug/g, lung concentrations of 0.75 and 23.71 ug/g, and brain concentrations of 0.044 and 0.710 ug/g. The ovary contained 1.16 ug/g and the testis contained 7.22 ug/g. A significant positive correlation existed between the administered dose and the concentration of PFOA in the liver, kidney, spleen, and lung of females. On the contrary, no significant correlation between the administered dose and the concentration of PFOA was observed in the males, as 10 mg/kg/day produced higher PFOA concentrations in the serum and organs than 30 mg/kg/day. However, in males, the concentration in the spleen, testis, and brain correlated positively with the concentration in the serum.

Vanden Heuvel et al. (1992) demonstrated that PFOA covalently binds to proteins in the liver, plasma, and testes of rats. Carbon-14-labeled PFOA was administered to six-week old male Harlan Sprague-Dawley rats in propylene glycol/water (1:1, v/v; 1 ml/kg) at a dose of 9.4 umol/kg, i.p. No time-dependent changes in either absolute or relative concentrations of covalently bound PFOA-derived ^{14}C were found at 2 h, 1 and 4 days post-treatment. Covalently bound PFOA was represented by 0.1 to 0.3% of the tissue ^{14}C content. The absolute concentration of covalently bound PFOA was significantly higher in the plasma than in the liver. The testes had the highest relative concentration of PFOA-derived radioactivity covalently bound. In tests, covalent binding of ^{14}C-PFOA to a constant concentration of albumin (8 uM) increased in a linear fashion with increasing PFOA concentration. The covalent binding of PFOA to hemoglobin was diminished by the addition of cysteine but not methionine, suggesting that protein sulfhydryl groups may be involved.

Hanhijarvi et al. (1987) compared the disposition of PFOA between male and female Wistar rats during subchronic administration. PFOA was administered by gavage to 48 newly-weaned animals at 0, 3, 10, and 30 mg/kg (in 0.9% NaCl, 0.5ml/100g) for 28 consecutive days. Urine was collected on the 7th and 28th day of the study (discussed below). At the end of the study, blood was collected via cardiac puncture. At each dose level, the mean PFOA concentrations in the plasma of the male rats were significantly higher than those of the female rats. The mean plasma PFOA concentrations for the male rats were 48.6+-26.5 ug/ml (dosed at 3 mg/kg), 83.1+-24.7 ug/ml (10 mg/kg), and 53.4+-11.2 ug/ml (30 mg/kg). The corresponding figures for female rats were 2.43+-5.96 ug/ml, 11.3+-8.59 ug/ml, and 9.06+-8.80 ug/ml in the same order. The PFOA concentrations in the plasma of the male animals suggested that the binding sites of PFOA may become saturated at the chronic daily dose level of 30 mg/kg. Although the plasma PFOA concentrations were significantly higher in the male rats, no significant histopathological differences between the sexes were observed at necropsy.

The disposition of PFOA was studied in male Wistar rats after castration and estradiol administration as well as in intact males and females (Ylinen et al., 1989). The male rats (N=20) were castrated at the age of 28 days and after 5 weeks were used in the tests. Half of the operated and 10 intact males were administered estradiol valerate subcutaneously 500 ug/kg every second day during 14 days before the test. Blood samples were collected by cardiac puncture. At the end of the test (96 hr), the concentration of PFOA in the serum of intact males was considerably higher (17-40 times) than in the serum of other groups. There was no statistically significant difference in the serum concentrations between the other groups. PFOA was similarly bound to the proteins in the serum of males and females.

Johnson et al. (1984) investigated the effect of feeding cholestyramine to rats on the fecal elimination of APFO. Since APFO exists as an anion at physiologic pH, it would be expected to complex with cholestyramine. Ten male Charles River CD rats (12 weeks old, 300-342 g) were administered ammonium ^{14}C-perfluorooctanoate (2.1 mg/ml) dissolved in 0.9% NaCl as a single intravenous dose (2 ml/rat, average APFO dose 13 mg/kg). Five rats were given 4% cholestyramine in feed. Urine and feces samples were collected at intervals for 14 days, at which time the animals were sacrificed and liver samples were collected. At 14 days post dose,

the mean percentage of PFOA dose eliminated in the feces of cholestyramine-treated rats (43.2+-5.5) was 9.8-fold the mean percentage of dose eliminated in feces by untreated rats (4.4+-1.0). Excretion in urine was 41% for treated rats and 67% for untreated rats. Carbon-14 present in the liver represented 12.1+-2.1 ug eq/g and 22.3+-6.2 ug eq/g in treated and untreated rats, respectively (4% and 8% of dose, respectively). In plasma, the levels were 5.1+-1.7 ug eq/ml and 14.7+-6.8 ug eq/ml in treated and untreated rats, respectively. In red blood cells, the levels were 1.8+-0.7 ug eq/ml and 4.2+-2.4 ug eq/ml in treated and untreated rats, respectively. The high concentration of ^{14}C-APFO in liver at 2 weeks after dosing and the fact that cholestyramine treatment enhances fecal elimination of carbon-14 nearly 10-fold suggests that there is enterohepatic circulation of PFOA.

The disposition of the tetrabutyl ammonium salt of perfluorooctanoic acid in female rabbits has been reported (Johnson, 1995a). Individual rabbits were given intravenous doses at 0, 4, 16, and 24 mg/kg and appeared normal throughout the study (the animal treated at the 40 mg/kg dose level died within 5 minutes of dosing). Serum samples were analyzed for total organic fluorine at 2, 4, 6, 8, 12, 24, and 48 hours post dose. At 2 hrs, serum organic fluorine levels in the 0, 4, 16, and 24 mg/kg dosed rabbits were 1.25 ppm, 4.09 ppm, 14.9 ppm, and 41.0 ppm, respectively. There was a rapid decrease in serum level of total organic fluorine with time, non-detectable at 48 hr. The biological half-life was on the order of 4 hours. The total organic fluorine in whole liver at 48 hr post dose for control animals, 4 mg/kg, 16 mg/kg, and 24 mg/kg intravenous doses were 20 ug, 43 ug, 66 ug, and 54 ug.

3.1.4 Metabolism Studies in Animals

Vanden Heuvel et al. (1991) investigated the metabolism of PFOA in rats administered ^{14}C-PFOA (9.4 umol/kg, i.p.). Pooled daily urine samples (0-4 days post-treatment) and bile extracts analyzed by HPLC contained a single radioactive peak eluting identically to the parent compound. Tissues were taken from rats treated 4, 14, and 28 days previously with ^{14}C-PFOA to determine the presence of PFOA-containing lipid conjugates. Only the parent compound was present in rat tissues; no PFOA-containing hybrid lipids were detected. Fluoride concentrations in plasma and urine before and after PFOA treatment were unchanged, indicating that PFOA does not undergo defluorination.

Ophaug and Singer (1980) also found no change in ionic fluoride level in the serum or urine following oral administration of PFOA to female rats. Ylinen et al. (1989) found no evidence of phase II metabolism of PFOA following a single intraperitoneal PFOA dose (50 mg/kg) in male and female rats.

3.1.5 Elimination Studies in Animals

There are major gender differences in the elimination of PFOA in rats. The biological half-life of PFOA in male rats is many times greater than that in female rats and this difference is primarily due to low renal clearance in male rats. The rapid excretion of PFOA by female rats is due to active renal tubular secretion (organic acid transport system); this renal tubular secretion

is believed to be hormonally controlled since castrated male rats treated with estradiol have excretion rates of PFOA similar to those of female rats. Hormonal changes during pregnancy do not appear to change the rate of elimination in rats. This gender difference has not been observed in primates and humans. The studies demonstrating this are described below.

The urine is the major route of excretion of PFOA in the female rat, while the urine and feces are both major routes of excretion of PFOA in male rats (Vanden Heuval, 1991). Male and female rats were administered ^{14}C-PFOA in propylene glycol/water (9.4 umol/kg, i.p.). Female rats eliminated PFOA-derived radioactivity rapidly in the urine with 91% of the dose being excreted in the first 24 hr, while male rats excreted only 6% of the dose in that time period. Negligible radioactivity was recovered in the feces of female rats. In male rats during the 28-day collection period the cumulative excretion of PFOA-derived ^{14}C in urine and feces was 36.4% and 35.1%, respectively. The female rat retained less than 10% of the administered dose after 24 hr, while the male rats retained 30% of the administered dose after 28 days. The whole-body elimination half-life in females was less than one day, and in males it was 15 days. In renal-ligated rats injected i.p. with ^{14}C-PFOA, approximately 0.3% of the PFOA-derived radioactivity was excreted in the bile after 6 hr (Vanden Heuvel et al., 1991). No sex-related difference in the biliary excretion of PFOA was observed when the kidneys were ligated.

Johnson and Gibson (1980) observed a sex difference in extent and rate of excretion of total carbon-14 between male and female rats after a single iv dose (mean dose: female, 16.7 mg/kg; male 13.1 mg/kg) of ^{14}C-PFOA. Female rats excreted essentially all of the dose via urine in 24 hours while at the same time period male rats excreted only 20 percent of the dose; male rats excreted 83% via urine and 5.4% via feces by 36 days post dose. No radioactivity was detected in tissues of female rats at 17 days post dose; male rats had 2.8% of the dose in liver and 1.1% in plasma at 36 days post dose with lower levels (< 0.5% of the dose) in other organs.

Ophaug and Singer (1980) investigated the metabolic fate of PFOA in female Holtzman rats. Animals weighing approximately 250 g were administered by stomach intubation 2 ml of an aqueous solution containing 2 mg PFOA. The animals were then placed in metabolism cages and provided rat chow and tap water for 4.5, 8, 24, or 52.5 hr. In addition, four rats were placed in metabolism cages and fed a low fluoride (<0.5 ppm) diet and distilled water for a period of 96 hr. At the end of the experimental period the urine, feces and serum were collected. Within 4.5 hr after PFOA dose, an average of 749 ug or 37% of the fluorine in the administered dose was recovered in the urine. The quantity of nonionic fluorine recovered in the urine increased to 61% of the dose at 8 hr, 76% at 24 hr, and 89% at 96 hr. Urinary excretion of ionic fluoride in the PFOA dosed animals was not significantly different than that of the control animals. Fecal excretion of nonionic fluorine was 4.5% of the administered dose at 52.5 hr and 14.3% at 96 hr. The urine from undosed animals contained no detectable nonionic fluorine.

The urinary excretion of APFO in rats was investigated by Hanhijarvi et al. (1982). Four male and six female Holtsman rats were administered 2 mg APFO in 2 ml aqueous solution by stomach intubation. Seven female rats were administered 2 ml distilled water as controls. The animals were then placed in metabolism cages with rat chow and tap water. Urine was collected

until animals were sacrificed at 24 h by cardiac puncture. Serum was collected. Ionic fluoride and total fluorine content of serum and urine was determined, and nonionic fluorine was calculated as the difference. For clearance studies of APFO and inulin, the rats were anesthetized with Inactin. The femoral artery was cannulated for continuous infusion of 5% mannitol in isotonic saline and the femoral artery was cannulated for drawing blood samples. The urinary bladder was also cannulated for serial collections of urine. Intravenous priming doses of 5.2-5.6 mg [1-14C] ammonium perfluorooctanoate (sp act 0.5 uCi/mg) and 8.8 ug tritiated inulin (methoxy-3H, sp act 114 uCi/mg) were given to each animal. The radiolabled inulin and APFO in 5% mannitol in isotonic saline was then infused at a rate of 0.21 ml/min. An additional 0.42-0.63 mg/hr ^{14}C-APFO and 9.6 ug/hr tritiated inulin was infused during the experiments. When the urine and serum collections for the clearance study were complete, probenecid was administered (65-68 mg/kg, ip) and additional clearance tests were performed. In the cumulative excretion study, rats were dosed iv with a mixture of radiolabeled APFO (10-20%) and unlabeled APFO (80-90%). Five percent mannitol in isotonic saline was infused at a rate of 0.081 ml/min and urine specimens were collected over 30-min intervals. The effect of probenecid was assessed by administering 65-68 mg/kg ip at least 30 min prior to the administration of APFO. Twenty-four hours after oral administration of APFO, female rats had excreted 76+-2.7% of the dose in the urine and had a mean serum nonionic fluorine level of 0.35+-0.11 ppm, while male rats had excreted only 9.2+-3.5% of the dose and had a mean serum nonionic fluorine level of 44.0+-1.7 ppm. APFO was bound to a similar extent in the plasma of male and female rats (97.5+-0.25% bound). The clearance studies demonstrated major differences between the sexes in rats. The APFO clearance in female rats was several times greater than the inulin clearance. Administration of probenecid, which strongly inhibits the renal active secretion of organic acids, reduced APFO/inulin clearance ratio in females from 14.5 to 0.46. APFO clearance was reduced from 5.8 to 0.11 ml/min/100g. Net APFO excretion was reduced from 4.6 to 0.13 ug/min/100g. In male rats, however, the APFO/inulin clearance ratio and the net excretion of APFO were virtually unaffected by probenecid. In the males, APFO clearance was 0.17 ml/min/100g, APFO/inulin clearance ratio was 0.22, and net APFO excretion was 0.17 ug/min/mg. In the cumulative excretion studies, female rats excreted 76% of the APFO dose, while males excreted only 7.8% of the dose over a 7-hr period. Probenecid administration modified the cumulative excretion curve for males only slightly. However, in females probenecid markedly reduced APFO elimination to 11.8%. The authors concluded that the female rat possesses an active secretory mechanism which rapidly eliminates APFO from the body. This secretory mechanism is lacking or is relatively inactive in male rats and accounts for the greater toxicity of APFO in male rats.

Hanhijarvi et al. (1987) compared the urinary elimination of PFOA between male and female Wistar rats during subchronic administration. APFO was administered by gavage to 48 newly-weaned animals at 0, 3, 10, and 30 mg/kg (in 0.9% NaCl, 0.5ml/100g) for 28 consecutive days. Urine was collected on the 7th and 28th day of the study. At the end of the study, blood was collected via cardiac puncture. At necropsy, tissue specimens for histopathologic examination were collected from the controls and from the group receiving 30 mg/kg/day PFOA. On the seventh day of the study period, the female rats in the lowest dose group (3 mg/kg/day) exhibited significantly greater urinary PFOA excretion than the males (3.12+-0.30 vs 1.50+-0.57

mg/24hr/kg). Unlike the female rats, on the 7th day of the study all three groups of male rats excreted significantly less PFOA than their daily dose of PFOA, which suggested that the males had not reached a steady state by seven days. On the 28th day, the males excreted an amount of PFOA equal to their daily dose.

Hanhijarvi et al. (1988) investigated the excretion kinetics of PFOA in the beagle dog. Six laboratory bred beagle dogs (3 male, 3 female) were anesthetized with methoxyflurane and catheters were placed in both ureters after laparototomy and cystotomy. The animals were given an intravenous dose of 30 mg/kg of PFOA followed by continuous infusion with 5% mannitol solution at 1.7 ml/min. Urine was collected at 10 minute intervals for 60 min. A 5 ml blood sample was collected in the middle of each urine sampling period. Probenicid (30 mg/kg i.v.) was then administered, and urine and blood samples were again collected as before. Renal clearance of PFOA was calculated for the before and after probenecid injection periods. Four additional dogs (2male, 2 female) were given 30 mg/kg PFOA intravenously. These dogs were kept in metabolism cages, and blood samples were collected intermittently for 30 days. Renal clearance rate was approximately 0.03 ml/min/kg. Probenecid significantly reduced the PFOA clearance in both sexes, indicating an active secretion mechanism for PFOA. The plasma half-life of PFOA was longer in the male dogs (473 h and 541 h) than in the female dogs (202 h and 305 h).

The urinary excretion of PFOA was studied in male Wistar rats after castration and estradiol administration as well as in intact males and females (Ylinen et al., 1989). The male rats (N=20) were castrated at the age of 28 days and after 5 weeks were used in the tests. Half of the operated and 10 intact males were administered estradiol valerate subcutaneously 500 ug/kg every second day during 14 days before the test. Urine was collected in metabolism cages during 96 hr after a single intraperitoneal PFOA dose (50 mg/kg). Blood samples were collected by cardiac puncture. Castration and administration of estradiol to the male rats had a significant stimulatory effect on the urinary excretion of PFOA. During the first 24 hours, female rats excreted 72+-5% (N=6) of the dose, whereas the intact males excreted only 9+-4% (N=6). After the estradiol treatment, both the intact and castrated males excreted PFOA in amounts similar to females (61+-19% and 68+-14%, respectively). The castrated males without estradiol treatment excreted PFOA in urine faster than the intact males (50+-13%), but less than the females and the estrogen treated males. At the end of the test (96 hr), the concentration of PFOA in the serum of intact males was considerably higher (17-40 times) than in the serum of other groups. There was no statistically significant difference in the serum concentrations between the other groups. PFOA was similarly bound by the proteins in the serum of males and females.

Vanden Heuvel et al. (1992a) investigated whether androgens or estrogens are involved in the marked sex-differences in the urinary excretion of PFOA. Castration of males greatly increased (> 1-fold) the elimination of ^{14}C-PFOA (9.4 umol/kg, i.p.) in urine, demonstrating that a factor produced by the testis is responsible for the slow elimination of PFOA in male rats. Castration plus 17B-estradiol had no further effect on PFOA elimination whereas castration plus testosterone replacement at the physiological level reduced PFOA elimination to the same level as rats with intact testis. Thus, in male rats, testosterone exerts an inhibitory effect on renal

excretion of PFOA. In female rats, neither ovariectomy or ovariectomy plus testosterone affected the urinary excretion of PFOA, demonstrating that the inhibitory effect of testosterone on PFOA renal excretion is a male-specific response. Probenecid, which inhibits the renal transport system, decreased the high rate of PFOA renal excretion in castrated males but had no effect on male rats with intact testis.

Hormonal changes during pregnancy do not appear to cause a change in the rate of elimination of ^{14}C after oral administration of a single dose of ammonium ^{14}C-PFOA (Gibson and Johnson, 1983). At 8 or 9 days after conception, four pregnant rats and 2 nonpregnant female rats were dosed (mean dose, 15 mg/kg) and individual urine samples were collected at 12, 24, 36, and 48 hours post dose and analyzed for ^{14}C content. Essentially all of the ^{14}C was eliminated via urine within 24 hours for both groups of rats.

Feeding of cholestyramine to rats enhanced the fecal elimination of APFO (Johnson et al. (1984). Male rats were administered APFO (2.1 mg/ml) dissolved in 0.9% NaCl as a single intravenous dose (2 ml/rat, average APFO dose 13 mg/kg). At 14 days post dose, the mean percentage of APFO dose eliminated in the feces of cholestyramine-treated rats (43.2+-5.5) was 9.8-fold the mean percentage of dose eliminated in feces by untreated rats (4.4+-1.0). Excretion in urine was 41% for treated rats and 67% for untreated rats.

Kudo et al. (2002) demonstrated in male and female rats that renal clearance (CL_R) of PFOA and the renal mRNA levels of specific organic anion transporters are markedly affected by sex hormones. The biological half-life of PFOA in male rats was found to be 70 times longer than in female rats and this difference is due primarily to low CL_R in male rats. Castration of male rats caused a 14-fold increase in CL_R of PFOA. The elevated PFOA CL_R in castrated males was reduced by treating them with testosterone. Treatment of male rats with estradiol increased the CL_R of PFOA. In female rats, ovariectomy caused a significant increase in CL_R of PFOA, which was reduced by estradiol treatment. Treatments of female rats with testosterone reduced the CL_R of PFOA. Treatment with probenecid, a known inhibitor of organic anion transporters, markedly reduced the CL_R of PFOA in male rats, castrated male rats, and female rats. To identify the transporter molecules that are responsible for PFOA transport in the rat kidney, renal mRNA levels of specific organic anion transporters were determined in male and female rats under various hormonal states and compared with the CL_R of PFOA. The level of OAT2 mRNA in male rats was only 13% that in female rats. Castration or estradiol treatment increased the level of OAT2 mRNA whereas treatment of castrated male rats with testosterone reduced it. Ovariectomy of female rats significantly increased the level of OAT3 mRNA. Multiple regression analysis of the data suggested that organic anion transporter 2 (OAT2) and OAT3 are responsible for urinary elimination of PFOA in the rat.

3.2 Epidemiology Studies

3.2.1 Medical Surveillance Studies from the Antwerp and Decatur Plants

A cross-sectional analysis of the data from the 2000 medical surveillance program at the Decatur

and Antwerp plants was undertaken to determine if there were any associations between PFOA and hematology, clinical chemistries, and hormonal parameters of volunteer employees (Olsen, et al., 2001e). The data were analyzed for all employees from both plant locations. Mean PFOA serum levels were 1.03 ppm for all male employees at the Antwerp plant and 1.90 ppm for all male employees at the Decatur plant. Male production employees at the Decatur plant had significantly higher ($p < .05$) mean serum levels (2.34 ppm) than those at the Antwerp plant (1.28 ppm). Non-production employees at both plants had mean levels below 1 ppm. PFOA serum levels were higher than the PFOS serum values at both plants, especially the Decatur plant where serum levels are higher overall. In addition, values for total organic fluorine were even higher than the PFOA levels.

Multivariable regression analyses were conducted to adjust for possible confounders that may affect the results of the clinical chemistry tests. The following variables were included: production job (yes or no), plant, age, body mass index (BMI), cigarettes/day, drinks/day and years worked at the plant. A positive significant association was reported between PFOA and cholesterol ($p = .05$) and PFOA and triglycerides ($p = .002$). Age was also significant in both analyses. Alcohol consumed per day was significant in the cholesterol model, while BMI and cigarettes smoked per day was significant for triglycerides. When both PFOA and PFOS were included in the analyses, neither reached statistical significance in the cholesterol model, while PFOA remained significant ($p = .02$) in the triglycerides model. HDL was negatively associated with PFOA ($p = .04$) and remained significant ($p = .04$) when both PFOA and PFOS were included in the model. A positive association ($p = .01$) between T3 and PFOA was also observed and remained statistically significant ($p = .05$) when PFOS was included in the model. BMI, cigarettes/day, alcohol/day were also significant in the model. None of the other clinical chemistry, thyroid or hematology measures were significantly associated with PFOA in the regression model.

A longitudinal analysis of the above data and previous medical surveillance results was performed to determine whether occupational exposure to fluorochemicals over time is related to changes in clinical chemistry and lipid results in employees of the Antwerp and Decatur facilities (Olsen, et al., 2001f). The clinical chemistries included: cholesterol, HDL, triglycerides, alkaline phosphatase, gamma glutamyl transferase (GGT), aspartate aminotransferase (AST), alanine aminotransferase (ALT), total and direct bilirubin. Medical surveillance data from 1995, 1997, and 2000 were analyzed using multivariable regression. The plants were analyzed using 3 subcohorts that included those who participated in 2 or more medical exams between 1995 and 2000. A total of 175 male employees voluntarily participated in the 2000 surveillance and at least one other. Only 41 employees were participants in all 3 surveillance periods.

When mean serum PFOA levels were compared by surveillance year, PFOA levels in the employees participating in medical surveillance at the Antwerp plant increased between 1994/95 and 1997 and then decreased slightly between 1997 and 2000. At the Decatur plant, PFOA serum levels decreased between 1994/95 and 1997 and then increased between 1997 and 2000. When analyzed using mixed model multivariable regression and combining Antwerp and Decatur employees, there was a statistically significant positive association between PFOA and

serum cholesterol (p = .0008) and triglycerides (p = .0002) over time. When analyzed by plant and also by subcohort, these associations were limited to the Antwerp employees (p = .005) and, in particular, the 21 Antwerp employees who participated in all 3 surveillance years (p = .001). However, the association between PFOA and triglycerides was also statistically significant (p = .02) for the subgroup in which employees participated in biomonitoring in 1994/95 and 2000. There was not a significant association between PFOA and triglycerides among Decatur workers. There were no significant associations between PFOA and changes over time in HDL, alkaline phosphatase, GGT, AST, ALT, total bilirubin, and direct bilirubin.

There are several limitations to the 2000 cross-sectional and longitudinal studies including: 1) serum PFOA levels were significantly higher at the Decatur plant than at the Antwerp plant, 2) all participants were volunteers, 3) there were several consistent differences in clinical chemistry profiles and demographics between employees of the Decatur and Antwerp plants (Antwerp employees as compared to Decatur employees had lower PFOA serum levels, were younger, had lower BMIs, worked fewer years, had higher alcohol consumption, higher mean HDL and bilirubin values, lower mean triglyceride, alkaline phosphatase, GGT, AST, and ALT values, and mean thyroid hormone values tended to be higher), 4) PFOS and other perfluorinated chemicals are also present in these plants, 5) in the cross-sectional study, plant populations cannot be compared because they were placed into quartiles based on PFOS serum distributions only which were different for each subgroup and not applicable to PFOA, 6) only one measurement at a certain point in time was collected for each clinical chemistry test, and 7) PFOA serum levels overall have been increasing over time in these employees. In addition, in the longitudinal study only a small number of employees participated in all 3 sampling periods (24%), different labs and analytical techniques for PFOA were used each year, and female employees could not be analyzed because of the small number of participants.

3.2.2 Medical Surveillance Studies from the Cottage Grove Plant

A voluntary medical surveillance program was offered to employees of the Cottage Grove, Minnesota plant in 1993, 1995, and 1997 (n = 111, 80 and 74 employees, respectively) (Olsen, et al., 1998b, Olsen et al., 2000). The clinical chemistry parameters (cholesterol, hepatic enzymes, and lipoprotein levels) used in the longitudinal and cross-sectional studies of the Antwerp and Decatur plants were also used in this study. In addition, in 1997 only, cholecystokinin-33 (CCK) was also measured at the Cottage Grove plant. CCK levels were observed because certain research has suggested that pancreas acinar cell adenomas seen in rats exposed to PFOA may be the result of increased CCK levels (Obourn, et al., 1997).

Only male employees involved in PFOA production were included in the study. Sixty-eight employees were common to the 1993 and 1995 sampling periods, 21 were common between 1995 and 1997, and 17 participated in all three surveillance years. Mean serum PFOA levels and ranges are provided in Table 2 of the Biomonitoring Section of this report. It should be noted that Cottage Grove has the highest serum PFOA levels of the 3 plants studied.

Employees' serum PFOA levels were stratified into 3 categories (<1, 1- <10, and ≥10 ppm),

chosen to provide a greater number of employees in the ≥ 10 ppm category. As employees' mean serum PFOA levels increased, no statistically significant abnormal liver function tests, hypolipidemia, or cholestasis were observed in any of the sampling years. Multivariable regression analyses controlling for potential confounders (age, alcohol consumption, BMI, and cigarettes smoked) yielded similar results. The authors also reported that renal function, blood glucose, and hematology measures were not associated with serum PFOA levels; however, these data were not provided in the paper.

The mean CCK value reported for the 1997 sample was 28.5 pg/ml (range 8.8 - 86.7 pg/ml). The means in the 2 serum categories < 10 ppm were at least 50% higher than in the ≥ 10 ppm category. A statistically significant ($p = .03$) negative association between mean CCK levels and the 3 PFOA serum categories was observed. A scatter plot of the natural log of CCK and PFOA shows that all but 2 CCK values are within the assay's reference range of 0 - 80 pg/ml. Both of these employees (CCK values of 80.5 and 86.7 pg/ml) had serum PFOA levels less than 10 ppm (0.6 and 5.6 ppm, respectively). A multiple regression model of the natural log of CCK and serum PFOA levels continued to display a negative association after adjusting for potential confounders.

The cross-sectional design is a limitation of this study. Only 17 subjects were common to all 3 sampling years. In addition, the medical surveillance program is a voluntary one. The participation rate of eligible production employees decreased from approximately 70% in 1993 to 50% in 1997. Also, the laboratory reference range changed substantially for ALT in 1997. Finally, different analytical methods were used to measure serum PFOA. Serum PFOA was determined by electrospray high-performance liquid chromatography/mass spectrometry in 1997, but by thermospray in 1993 and 1995.

An earlier medical surveillance study on workers who were employed in the 1980's was conducted at the Cottage Grove plant; however, total serum fluorine was measured instead of PFOA (Gilliland and Mandel, 1996). Based on animal studies that reported that animals exposed to PFOA develop hepatomegaly and alterations in lipid metabolism, a cross-sectional, occupational study was performed to determine if similar effects are present in workers exposed to PFOA. In a PFOA production facility, 115 workers were studied to determine whether serum PFOA affected their cholesterol, lipoproteins, and hepatic enzymes. Forty-eight workers who were exposed to PFOA from 1985-1989 were included in the study (96% participation rate). Sixty-five employees who either volunteered or were asked to participate, were included in the unexposed group. These employees were assumed to have little or no PFOA exposure based on their job description. However, when serum levels were analyzed, it was noted that this group of workers had PFOA levels much greater than the general population. Therefore, instead of job categories, total serum fluorine was used to classify workers into exposure groups.

Total serum fluorine was used as a surrogate measure for PFOA. Serum PFOA was not measured, due to the cost of analyzing the samples. Blood samples were analyzed for total serum fluorine, serum glutamyl oxaloacetic transaminase (SGOT or AST), serum glutamyl pyruvic transaminase (SGPT or ALT), gamma glutamyl transferase (GGT), cholesterol, low-

density lipoproteins (LDL), and high-density lipoproteins (HDL). All of the participants were placed into five categories of total serum fluorine levels: <1 ppm, 1-3 ppm, >3 - 10 ppm, >10 - 15 ppm, and > 15 ppm. The range of the serum fluorine values was 0 to 26 ppm (mean 3.3 ppm). Approximately half of the workers fell into the > 1 - 3 ppm category, while 23 had serum levels < 1 ppm and 11 had levels > 10 ppm.

There were no significant differences between exposure categories when analyzed using univariate analyses for cholesterol, LDL, and HDL. In the multivariate analysis, there was not a significant association between total serum fluorine and cholesterol or LDL after adjusting for alcohol consumption, age, BMI, and cigarette smoking. There were no statistically significant differences among the exposure categories of total serum fluorine for AST, ALT and GGT. However, increases in AST and ALT occurred with increasing total serum fluorine levels in obese workers (BMI = 35 kg/m^2). This result was not observed when PFOA was measured directly in serum of workers in 1993, 1995, or 1997 surveillance data of employees of the Cottage Grove plant (Olsen, et al., 2000).

Since PFOA was not measured directly and there is no exposure information provided on the employees (eg. length of employment/exposure), the results of the study provide limited information. The authors state that no adverse clinical outcomes related to PFOA exposure have been observed in these employees; however, it is not clear that there has been follow-up of former employees. In addition, the range of results reported for the liver enzymes were fairly wide for many of the exposure categories, indicating variability in the results. Given that only one sample was taken from each employee, this is not surprising. It would be much more helpful to have several samples taken over time to ensure their reliability. It also would have been interesting to compare the results of the workers who were known to be exposed to PFOA to those who were originally thought not to be exposed to see if there were any differences among the employees in these groups. There were more of the "unexposed" employees (n = 65) participating in the study than those who worked in PFOA production (n = 48).

3.2.3 Mortality Studies

A retrospective cohort mortality study was performed on employees at the Cottage Grove, MN plant which produces APFO (Gilliland and Mandel, 1993). At this plant, APFO production was limited to the Chemical Division. The cohort consisted of workers who had been employed at the plant for at least 6 months between January 1947 and December 1983. Death certificates of all of the workers were obtained to determine cause of death. There was almost complete follow-up (99.5%) of all of the study participants. The exposure status of the workers was categorized based on their job histories. If they had been employed for at least 1 month in the Chemical Division, they were considered exposed. All others were considered to be not exposed to PFOA. The number of months employed in the Chemical Division provided the cumulative exposure measurements. Of the 3537 (2788 men and 749 women) employees who participated in this study, 398 (348 men and 50 women) were deceased. Eleven of the 50 women and 148 of the 348 men worked in the Chemical Division, and therefore, were considered exposed to PFOA.

Standardized Mortality Ratios (SMRs), adjusted for age, sex, and race were calculated and compared to U.S. and Minnesota white death rates for men. For women, only state rates were available. The SMRs for males were stratified for 3 latency periods (10, 15, and 20 years) and 3 periods of duration of employment (5, 10, and 20 years).

For all female employees, the SMRs for all causes and for all cancers were less than 1. The only elevated (although not significant) SMR was for lymphopoietic cancer, and was based on only 3 deaths. When exposure status was considered, SMRs for all causes of death and for all cancers were significantly lower than expected, based on the U.S. rates, for both the Chemical Division workers and the other employees of the plant.

In all male workers at the plant, the SMRs were close to 1 for most of the causes of death when compared to both the U.S. and the Minnesota death rates. When latency and duration of employment were considered, there were no elevated SMRs. When employee deaths in the Chemical Division were compared to Minnesota death rates, the SMR for prostate cancer for workers in the Chemical Division was 2.03 (95% CI .55 - 4.59). This was based on 4 deaths (1.97 expected). There was also a statistically significant association with length of employment in the Chemical Division and prostate cancer mortality. Based on the results of proportional hazard models, the relative risk for a 1-year increase in employment in the Chemical Division was 1.13 (95% CI 1.01 to 1.27). It rose to 3.3 (95% CI 1.02 -10.6) for workers employed in the Chemical Division for 10 years when compared to the other employees in the plant. The SMR for workers not employed in the Chemical Division was less than expected for prostate cancer (.58).

An update of this study was conducted to include the death experience of employees through 1997 (Alexander, 2001a). The cohort consisted of 3992 workers. The eligibility requirement was increased to 1 year of employment at the Cottage Grove plant, and the exposure categories were changed to be more specific. Workers were placed into 3 exposure groups based on job history information: definite PFOA exposure (n = 492, jobs where cell generation, drying, shipping and packaging of PFOA occurred throughout the history of the plant); probable PFOA exposure (n = 1685, other chemical division jobs where exposure to PFOA was possible but with lower or transient exposures); and not exposed to fluorochemicals (n = 1815, primarily non-chemical division jobs).

In this new cohort, 607 deaths were identified: 46 of these deaths were in the PFOA exposure group, 267 in the probable exposure group, and 294 in the non-exposed group. When all employees were compared to the state mortality rates, SMRs were less than 1 or only slightly higher for all of the causes of death analyzed. None of the SMRs were statistically significant at $p = .05$. The highest SMR reported was for bladder cancer (SMR = 1.31, 95% CI = 0.42 – 3.05). Five deaths were observed (3.83 expected).

A few SMRs were elevated for employees in the definite PFOA exposure group: 2 deaths from cancer of the large intestine (SMR = 1.67, 95% CI = 0.02 – 6.02), 1 from pancreatic cancer

(SMR = 1.34, 95% CI = 0.03 – 7.42), and 1 from prostate cancer (SMR = 1.30, 95% CI = 0.03 – 7.20). In addition, employees in the definite PFOA exposure group were 2.5 times more likely to die from cerebrovascular disease (5 deaths observed, 1.94 expected; 95% CI = 0.84 – 6.03).

In the probable exposure group, 3 SMRs should be noted: cancer of the testis and other male genital organs (SMR = 2.75, 95% CI = 0.07 – 15.3); pancreatic cancer (SMR = 1.24, 95% CI = 0.45 – 2.70); and malignant melanoma of the skin (SMR = 1.42, 95% CI = 0.17 – 5.11). Only 1, 6, and 2 cases were observed, respectively. The SMR for prostate cancer in this group was 0.86 (95% CI = 0.28 – 2.02) (n = 5).

There were no notable excesses in SMRs in the non-exposed group, except for cancer of the bladder and other urinary organs. Four cases were observed and only 1.89 were expected (95% CI = 0.58 – 5.40).

It is difficult to interpret the results of the prostate cancer deaths between the first study and the update because the exposure categories were modified in the update. Only 1 death was reported in the definite exposure group and 5 were observed in the probable exposure group. All of these deaths would have been placed in the chemical plant employees exposure group in the first study. The number of years that these employees worked at the plant and/or were exposed to PFOA was not reported. This is important because even 1 prostate cancer death in the definite PFOA exposure group resulted in an elevated SMR for the group. Therefore, if any of the employees' exposures were misclassified, the results of the analysis could be altered significantly.

The excess mortality in cerebrovascular disease noted in employees in the definite exposure group was further analyzed based on number of years of employment at the plant. Three of the 5 deaths occurred in workers who were employed in jobs with definite PFOA exposure for more than 5 years but less than 10 years (SMR = 15.03, 95% CI = 3.02 – 43.91). The other 2 occurred in employees with less than 1 year of definite exposure. The SMR was 6.9 (95% CI = 1.39 – 20.24) for employees with greater than 5 years of definite PFOA exposure. In order to confirm that the results regarding cerebrovascular disease were not an artifact of death certificate coding, regional mortality rates were used for the reference population. The results did not change. When these deaths were further analyzed by cumulative exposure (time-weighted according to exposure category), workers with 27 years of exposure in probable PFOA exposed jobs or those with 9 years of definite PFOA exposure were 3.3 times more likely to die of cerebrovascular disease than the general population. A dose-response relationship was not observed with years of exposure.

It is difficult to compare the results of the first and second mortality studies at the Cottage Grove plant since the exposure categories were modified. Although the potential for exposure misclassification was certainly more likely in the first study, it may still have occurred in the update as well. It is difficult to judge the reliability of the exposure categories that were defined without measured exposures. Although serum PFOA measurements were considered in the exposure matrix developed for the update, they were not directly used. In the second study, the

chemical plant employees were sub-divided into PFOA-exposed groups, and the film plant employees essentially remained in the "non-exposed" group. This was an effort to more accurately classify exposures; however, these new categories do not take into account duration of exposure or length of employment. Another limitation to this study is that 17 death certificates were not located for deceased employees and therefore were not included in the study. The inclusion or exclusion of these deaths could change the analyses for the causes of death that had a small number of cases. Follow up of worker mortality at Cottage Grove (and Decatur) needs to continue. Although there were more than 200 additional deaths included in this analysis, it is a small number and the cohort is still relatively young. Given the results of studies on fluorochemicals in both animals and humans, further analysis is warranted.

3.2.4 Hormone Study

Endocrine effects have been associated with PFOA exposure in animals; therefore, medical surveillance data, including hormone testing, from employees of the Cottage Grove, Minnesota plant were analyzed (Olsen, et al., 1998a). PFOA serum levels were obtained for volunteer workers in 1993 (n = 111) and 1995 (n = 80). Sixty-eight employees were common to both sampling periods. In 1993, the range of PFOA was 0-80 ppm (although 80 ppm was the limit of detection that year, so it could have been higher) and 0-115 ppm in 1995 using thermospray mass spectrophotometry assay. Eleven hormones were assayed from the serum samples. They were: cortisol, dehydroepiandrosterone sulfate (DHEAS), estradiol, FSH, 17 gamma-hydroxyprogesterone (17-HP), free testosterone, total testosterone, LH, prolactin, thyroid-stimulating hormone (TSH) and sex hormone-binding globulin (SHBG). Employees were placed into 4 exposure categories based on their serum PFOA levels: 0-1 ppm, 1- < 10 ppm, 10- < 30 ppm, and >30 ppm. Statistical methods used to compare PFOA levels and hormone values included: multivariable regression analysis, ANOVA, and Pearson correlation coefficients.

PFOA was not highly correlated with any of the hormones or with the following covariates: age, alcohol consumption, BMI, or cigarettes. Most of the employees had PFOA serum levels less than 10 ppm. In 1993, only 12 employees had serum levels > 10 ppm, and 15 in 1995. However, these levels ranged from approximately 10 ppm to over 114 ppm. There were only 4 employees in the >30 ppm PFOA group in 1993 and only 5 in 1995. Therefore, it is likely that there was not enough power to detect differences in either of the highest categories. The mean age of the employees in the highest exposure category was the lowest in both 1993 and 1995 (33.3 years and 38.2 years, respectively). Although not significantly different from the other categories, BMI was slightly higher in the highest PFOA category.

Estradiol was highly correlated with BMI ($r = .41$, $p < .001$ in 1993, and $r = .30$, $p < .01$ in 1995). In 1995, all 5 employees with PFOA levels > 30 ppm had BMIs > 28, although this effect was not observed in 1993. Estradiol levels in the >30 ppm group in both years were 10% higher than the other PFOA groups; however, the difference was not statistically significant. The authors postulate that the study may not have been sensitive enough to detect an association between PFOA and estradiol because measured serum PFOA levels were likely below the observable effect levels suggested in animal studies (55 ppm PFOA in the CD rat). Only 3

employees in this study had PFOA serum levels this high. They also suggest that the higher estradiol levels in the highest exposure category could suggest a threshold relationship between PFOA and estradiol.

Free testosterone was highly correlated with age in both 1993 and 1995. The authors did not report a negative association between PFOA serum levels and testosterone. There were no statistically significant trends noted for PFOA and either bound or free testosterone. However, 17-HP, a precursor of testosterone, was highest in the >30 ppm PFOA group in both 1993 and 1995. In 1995, PFOA was significantly associated with 17-HP in regression models adjusted for possible confounders. However, the authors state that this association was based on the results of one employee (data were not provided in the report). There were no significant associations between PFOA and cortisol, DHEAS, FSH, LH, and SHBG.

There are several design issues that should be noted when evaluating the results of this study. First, although there were 2 study years (1993 and 1995), the populations were not independent. Sixty-eight employees participated in both years. Second, there were 31 fewer employees who participated in the study in 1995, thus reducing the power of the study. There were also very few employees in either year with serum PFOA levels greater than 10 ppm. Third, the cross-sectional design of the study does not allow for analysis of temporality of an association. Since the half-life of PFOA is at least 1 year, the authors suggest that it is possible that there may be some biological accommodation to the effects of PFOA. Fourth, only one sample was taken for each hormone for each of the study years. In order to get more accurate measurements for some of the hormones, pooled blood taken in a short time period should have been used for each participant. Fifth, some of the associations that were measured in this study were done based on the results of an earlier paper that linked PFOA with increased estradiol and decreased testosterone levels. However, total serum organic fluorine was measured in that study instead of PFOA, making it difficult to compare the results. Finally, there may have been some measurement error of some of the confounding variables.

3.2.5 Study on Episodes of Care (Morbidity)

In order to gain additional insight into the effects of fluorochemical exposure on workers' health, an "episode of care" analysis was undertaken at the Decatur plant to screen for morbidity outcomes that may be associated with long-term, high exposure to fluorochemicals (Olsen et al., 2001g). An "episode of care" is a series of health care services provided from the start of a particular disease or condition until solution or resolution of that problem. Episodes of care were identified in employees' health claims records using Clinical Care Groups (CCG) software. All inpatient and outpatient visits to health care providers, procedures, ancillary services and prescription drugs used in the diagnosis, treatment, and management of over 400 diseases or conditions were tracked.

Episodes of care were analyzed for 652 chemical employees and 659 film plant employees who worked at the Decatur plant for at least 1 year between January 1, 1993 and December 31, 1998. Based on work history records, employees were placed into different comparison groups:

Group A consisted of all film and chemical plant workers; Group B had employees who only worked in either the film or chemical plant; Group C consisted of employees who worked in jobs with high POSF exposures; and Group D had employees who worked in high exposures in the chemical plant for 10 years or more prior to the onset of the study. Film plant employees were considered to have little or no fluorochemical exposure, while chemical plant employees were assumed to have the highest exposures.

Ratios of observed to expected episodes of care were calculated for each plant. Expected numbers were based on 3M's employee population experience using indirect standardization techniques. A ratio of the chemical plant's observed to expected experience divided by the film plant's observed to expected experience was calculated to provide a relative risk ratio for each episode of care (RREpC). 95% confidence intervals were calculated for each RREpC. Episodes of care that were of greatest interest were those which had been reported in animal or epidemiologic literature on PFOS and PFOA: liver and bladder cancer, endocrine disorders involving the thyroid gland and lipid metabolism, disorders of the liver and biliary tract, and reproductive disorders.

The only increased risk of episodes for these conditions of a priori interest were for neoplasms of the male reproductive system and for the overall category of cancers and benign growths (which included cancer of the male reproductive system). There was an increased risk of episodes for the overall cancer category for all 4 comparison groups. The risk ratio was greatest in the group of employees with the highest and longest exposures to fluorochemicals (RREpC = 1.6, 95% CI = 1.2 – 2.1). Increased risk of episodes in long-time, high-exposure employees also was reported for male reproductive cancers (RREpC = 9.7, 95% CI = 1.1 - 458). It should be noted that the confidence interval is very wide for male reproductive cancers and the sub-category of prostate cancer. Five episodes of care were observed for reproductive cancers in chemical plant employees (1.8 expected), of which 4 were prostate cancers. One episode of prostate cancer was observed in film plant employees (3.4 expected). This finding should be noted because an excess in prostate cancer mortality was observed in the Cottage Grove plant mortality study when there were only 2 exposure categories (chemical division employees and non-chemical division employees). The update of the study sub-divided the chemical plant employees and did not confirm this finding when exposures were divided into definitely exposed and probably exposed employees.

There was an increased risk of episodes for neoplasms of the gastrointestinal tract in the high exposure group (RREpC = 1.8, 95% CI = 1.2- 3.0) and the long-term employment, high exposure group (RREpC = 2.9, 95% CI = 1.7 – 5.2). Most of the episodes were attributable to benign colonic polyps. Similar numbers of episodes were reported in film and chemical plant employees.

In the entire cohort, only 1 episode of care was reported for liver cancer (0.6 expected) and 1 for bladder cancer (1.5 expected). Both occurred in film plant employees. Only 2 cases of cirrhosis of the liver were observed (0.9 expected), both in the chemical plant. There was a greater risk of lower urinary tract infections in chemical plant employees, but they were mostly due to recurring

episodes of care by the same employees. It is difficult to draw any conclusions about these observations, given the small number of episodes reported.

Chemical plant employees in the high exposure, long-term employment group were 2 ½ times more likely to seek care for disorders of the biliary tract than their counterparts in the film plant (RREpC = 2.6, 95% CI = 1.2 - 5.5). Eighteen episodes of care were observed in chemical plant employees and 14 in film plant workers. The sub-categories that influenced this observation were episodes of cholelithiasis with acute cholecystitis and cholelithiasis with chronic or unspecified cholecystitis. Most of the observed cases occurred in chemical plant employees.

Risk ratios of episodes of care for endocrine disorders, which included sub-categories of thyroid disease, diabetes, hyperlipidemia, and other endocrine or nutritional disorders, were not elevated in the comparison groups. Conditions which were not identified a priori but which excluded the null hypothesis in the 95% confidence interval for the high exposure, long-term employment group included: disorders of the pancreas, cystitis, and lower urinary tract infections.

The results of this study only should be used for hypothesis generation. Although the episode of care design allowed for a direct comparison of workers with similar demographics but different exposures, there are many limitations to this design. The limitations include: 1) episodes of care are reported, not disease incidence, 2) the data are difficult to interpret because a large RREpC may not necessarily indicate high risk of incidence of disease, 3) many of the risk ratios for episodes of care had very wide confidence intervals, thereby providing unstable results, 4) the analysis was limited to 6 years, 5) the utilization of health care services may reflect local medical practice patterns, 6) individuals may be counted more than once in the database because they can be categorized under larger or smaller disease classifications, 7) episodes of care may include the same individual several times, 8) not all employees were included in the database, such as those on long-term disability, 9) the analysis may be limited by the software used, which may misclassify episodes of care, 10) the software may assign 2 different diagnoses to the same episode, and 11) certain services, such as lab procedures may not have been reported in the database.

3.3 Acute Toxicity Studies in Animals

3.3.1 Oral Studies

The acute oral toxicity of APFO was tested in male and female rats in three studies. Death occurred at concentrations ≥464 mg/kg (Internat'l Res and Dev Corp., 1978). Abnormal findings upon necropsy (kidney, stomach, uterus) were observed (Glaza,1997) at 500 mg/kg (higher concentrations were not tested). Clinical signs of toxicity observed in these three studies included the following: red-stained face, stained urogenital area, wet urogenital area, hypoactivity, hunched posture, staggered gait, excessive salivation, ptosis, piloerection, decreased limb tone, ataxia, corneal opacity, and hypothermic to touch.

In one study (Internat'l Res and Dev Corp., 1978), the oral LD50 values for Charles River CD

rats were 680 mg/kg (399 – 1157 mg/kg 95% confidence limit) for males; 430 mg/kg (295 – 626 mg/kg 95% confidence limit) for females; and 540 mg/kg (389 – 749 mg/kg 95% confidence limit) for males and females. The remaining two studies provided LD50 values of (1) >500 mg/kg for male Crl:CD(SD)BR rats, and 250-500 mg/kg for female Crl:CD(SD)BR rats (Glaza,1997); and (2) <1000 mg/kg for male and female Sherman-Wistar rats (3M Company, 1976b).

3.3.2 Inhalation Studies

The acute inhalation toxicity of APFO was tested in male and female Sprague-Dawley rats, at a dose level of 18.6 mg/L (nominal concentration), and exposure duration of one hour. Signs of toxicity during and up to 14 days after the exposure period, included the following: excessive salivation, excessive lacrimation, decreased activity, labored breathing, gasping, closed eyes, mucoid nasal discharge, irregular breathing, red nasal discharge, yellow staining of the anogenital fur, dry and moist rales, red material around the eyes, and body tremors. Upon necropsy, lung discoloration was observed in a higher than normal incidence of rats (8/10). Based on the study results, the test substance was not fatal to rats at a nominal exposure concentration of 18.6 mg/L and exposure duration of one hour (Bio/dynamics, Inc. 1979).

3.3.3 Dermal Studies

The acute dermal toxicity of APFO was tested in male and female Hra(NZW)SPF rabbits, at a dose level of 2000 mg/kg, and a 24-hour exposure period. All animals appeared normal and exhibited body weight gain throughout the study, with the exception of one male that lost weight during the first week. Dermal irritation consisted of slight to moderate erythema, edema, and atonia; slight desquamation; coriaceousness; and fissuring. No visible lesions were observed upon necropsy. The dermal LD50 in rabbits was determined to be greater than 2000 mg/kg (Glaza, 1995).

3.3.4 Eye Irritation Studies

The eye irritation potential of APFO was tested in albino rabbits, at a dose level of 0.1 gram. In two of three studies, APFO was determined to be a primary ocular irritant. In the studies in which APFO was found to be a primary ocular irritant, APFO was left in contact with the eye for 7 days, then rinsed, or not rinsed. Irritation scores varied during the observation period. Irritation scores of the conjunctivae, iris, and cornea ranged from 2 – 4 in one study (Biosearch, Inc. 1976) and from 2 –10 in the other study (3M Company, 1976a). In both studies, irritation remained evident for the duration of the observation period (7-days post-exposure). In the study in which APFO was determined to be a non-irritant (Gabriel), the test substance was left in contact with the eye for 5 or 30 seconds, and then the eyes were rinsed. In this study, positive scores were reported for conjunctivae irritation for up to 7-days post-exposure, so the author's negative conclusion for ocular irritancy is problematic.

3.3.5 Skin Irritation Studies

The skin irritation potential of APFO was tested in albino rabbits in two studies, at a dose level of 0.5 grams, under occluded test conditions. In one study (Riker Laboratories, Inc. 1983), APFO produced irreversible tissue damage in female rabbits, following a 3-minute, 1-hour, and 4-hour contact period. Moderate erythema and edema, as well as chemical burn, eschar, and necrosis, were observed following all three contact periods. An endpoint was not achieved in this study due to extreme irritation following each contact period. In contrast in the second study (Gabriel), APFO was reported as a non-irritant of skin after an exposure period of 24 or 72 hours, based on primary irritation scores of zero.

3.4 Mutagenicity Studies

APFO was tested twice (Lawlor, 1995; 1996) for its ability to induce mutation in the *Salmonella – E. coli*/mammalian-microsome reverse mutation assay. The tests were performed both with and without metabolic activation. A single positive response seen at one dose level in S. typhimurium TA1537 when tested without metabolic activation was not reproducible. APFO did not induce mutation in either S. typhimurium or E. coli when tested either with or without mammalian activation. APFO did not induce chromosomal aberrations in human lymphocytes when tested with and without metabolic activation up to cytotoxic concentrations (Murli, 1996c; NOTOX, 2000). Sadhu (2002) recently reported that APFO did not induce gene mutation when tested with or without metabolic activation in the K-1 line of Chinese hamster ovary (CHO) cells in culture.

Murli (1996b) tested APFO twice for its ability to induce chromosomal aberrations in CHO cells. In the first assay, APFO induced both chromosomal aberrations and polyploidy in both the presence and absence of metabolic activation. In the second assay, no significant increases in chromosomal aberrations were observed without activation. However, when tested with metabolic activation, APFO induced significant increases in chromosomal aberrations and in polyploidy (Murli, 1996b).

APFO was tested in a cell transformation and cytotoxicity assay conducted in C_3H $10T_{½}$ mouse embryo fibroblasts. The cell transformation was determined as both colony transformation and foci transformation potential. There was no evidence of transformation at any of the dose levels tested in either the colony or foci assay methods (Garry & Nelson, 1981).

APFO was tested twice in the mouse micronucleus assay. APFO did not induce any significant increases in micronuclei and was considered negative under the conditions of this assay (Murli, 1996a).

3.5 Subchronic Toxicity Studies in Animals

Subchronic toxicity studies have been conducted in rats, mice, rhesus monkeys and cynomolgus monkeys. Two unpublished 28-day feeding studies were performed at Industrial Bio-Test

Laboratories, Inc. (Metrick and Marias, 1977 and Christopher and Marias, 1977). In both rats and mice the liver was the target organ. In rats, males had more pronounced hepatotoxicity and histopathologic effects than females. Three 90-day subchronic toxicity studies have been conducted in rats (Goldenthal, 1978a, Palazzolo, 1993) and rhesus monkeys (Goldenthal, 1978b). Thomford (2001a,b) recently conducted a range-finding and a 6-month toxicity study in cynomolgus monkeys. In addition, chronic/carcinogenicity studies have been conducted which are described in section 3.8. In all species, the liver is the main target organ. In rats, males had more pronounced hepatotoxicity and histopathologic effects than females, presumably because of the gender difference in elimination of APFO.

In a 28-day study of ChR-CD albino rats, eight randomly assigned groups of five males and five females were studied (Metrick and Marias, 1977). After rats were allowed to acclimate for a week in individual cages they then received similar feed containing 0, 30, 100, 300, 1000, 3000, 10,000, or 30,000 ppm APFO for 28 days. The animals were observed daily and body weights and food consumption were recorded weekly. Animals that died during the study were examined for gross pathology, as were surviving animals at 28 days. It is stated that the study included a complete examination of gross pathology and a complete set of tissues and organs were examined, but the specific list is not supplied. Livers were weighed to determine relative organ weight then stained for histopathologic examination.

All animals in the 10,000 and 30,000-ppm groups died before the end of the first week. There were no premature deaths or other clinical signs of toxicity in the other groups. Body weight gains were reduced in the groups receiving 1000 or more ppm. Slight reductions in body weight gain were also observed in males exposed to 300 ppm and males and females fed 100 ppm. Reduced food intake was observed in rats fed 1000 ppm or higher in a dose-related manner. Relative liver weights were increased in males fed 30 ppm or more and females fed 300 ppm or more. Gross pathological exam did not reveal treatment-related effects in kidneys or other organs besides livers. Focal to multifocal cytoplasmic enlargement of hepatocytes was noted in animals fed 300 ppm, and multifocal to diffuse enlargement of hepatocytes was noted in animals fed 1000 ppm or higher. These effects were more pronounced in males (Metrick and Marias. 1977).

In a 28-day study of Charles River-CD albino mice, eight randomly assigned groups of five males and five females were studied (Christopher and Marisa, 1977). After mice were allowed to acclimate for 8 days in individual cages they then received similar feed containing 0, 30, 100, 300, 1000, 3000, 10,000, or 30,000 ppm of APFO for 28 days. The animals were observed daily and body weights and food consumption were recorded weekly. Animals that died during the study were examined for gross pathology, as were surviving animals at 28 days. It is stated the study included a complete examination of gross pathology and a representative set of tissues and organs were examined, but the specific list is not supplied. Livers were weighed to determine relative organ weight then stained for histopathologic examination.

All animals in the 1000-ppm and higher groups died before the end of day 9. The entire 300-ppm group died within 26 days except 1 male. One animal in each of the 30 and 100-ppm groups

died prematurely. Clinical signs were observed in mice exposed to 100 ppm and higher doses of PFOA. At 100 ppm some animals exhibited cyanosis on days 10 and 11 of testing, but appeared normal throughout the rest of the study. Animals feed 300 ppm exhibited roughed fur and muscular weakness as well as signs of cyanosis after 9 days of treatment. Animals fed 1000 ppm exhibited similar effects after 6 days and those receiving 3000 ppm or greater doses exhibited effects after 4 days.

All mice fed APFO lost weight. Reductions in body weight gain were followed by weight losses in mice fed 30, 100, or 300 ppm. A dose-related pattern was seen in the depressed body weights.

Relative and absolute liver weights were increased in mice fed 30 ppm or more APFO. Gross pathological examination of kidneys or other organs besides livers is not discussed. Treatment-related changes were observed in the livers among all APFO treated animals including enlargement and/or discoloration of 1 or more liver lobes. Histopathologic examination of all APFO treated mice revealed diffuse cytoplasmic enlargement of hepatocytes throughout the liver (pan lobular hypertrophy) accompanied by focal to multifocal cytoplasmic vacuoles. Degeneration and /or necrosis of hepatocytes and focal bile duct proliferation were also noted in mice within all groups (Christopher and Marias, 1977).

Three 90-day subchronic toxicity studies have been conducted. One was conducted in rats (Goldenthal, 1978a), one was conducted in rhesus monkeys (Goldenthal, 1978b) and the third was conducted in male rats (Palazzolo, 1993).

In the monkey study, Goldenthal (1978b) administered rhesus monkeys (2/sex/group) doses of 0, 3, 10, 30 or 100 mg/kg/day perfluorooctanoic acid (FC-143) in 0.5% Methocel7 by gavage for 7 days/week for 90 days. All doses were given in a constant volume; individual daily doses were based upon the weekly body weights. Animals were observed twice daily for general physical appearance and behavior and pharmacotoxic signs. General physical examinations were performed during the control period and monthly during the study period. Individual body weights were recorded weekly. Blood and urine samples were collected once during the control period and at 1 and 3 months of the study for hematology, clinical chemistry and urinalysis. Monkeys were fasted overnight prior to the collection of blood and urine samples. Organs and tissues from animals that were sacrificed at the end of the study and from animals that died during the treatment period were weighed, examined for gross pathology and samples taken for histopathology. Histopathology was performed on the following organs from all monkeys in the control and treatment groups: adrenals, aorta, bone, brain, esophagus, eyes, gallbladder, heart (with coronary vessels), duodenum, ileum, jejunum, cecum, colon, rectum, kidneys, liver, lung, skin, mesenteric lymph node, retropharyngeal lymph node, mammary gland, nerve (with muscle), spleen, pancreas, prostate/uterus, rib junction (bone marrow), salivary gland, lumbar spinal cord, pituitary, stomach, testes/ovaries, thyroid, parathyroid, thymus, trachea, tonsil, tongue, urinary bladder, vagina, identifying tattoo, and any tissues(s) with lesions.

All monkeys in the 100-mg/kg/day groups died during the study. The first death occurred during week 2; all animals were dead by week 5. Signs and symptoms which first appeared during

week 1 included anorexia, frothy emesis which was sometimes brown in color, pale face and gums, swollen face and eyes, slight to severe decreased activity, prostration and body trembling. Three monkeys from the 30-mg/kg/day group died during the study; one male died during week 7 and the two females died during weeks 12 and 13. Beginning in week 4, all four animals showed slight to moderate and sometimes-severe decreased activity. One monkey had emesis and ataxia, swollen face, eyes and vulva, as well as pallor of the face and gums. Beginning in week 6, two monkeys had black stools and one monkey had slight to moderate dehydration and ptosis of the eyelids.

No monkeys in the 3 or 10 mg/kg/day groups died during the study. Animals in the 3-mg/kg/day-dose group occasionally had soft stools or moderate to marked diarrhea; frothy emesis was also occasionally noted in this group. One monkey in the 10 mg/kg/day group was anorexic during week 4, had a pale and swollen face in week 7 and had black stools for several days in week 12. The other animals in the 10-mg/kg/day groups did not show any unusual signs or symptoms.

Changes in body weight were similar to the controls for animals from the 3 and 10 mg/kg/day dose groups. Monkeys from the 30 and 100 mg/kg/day groups lost body weight after week 1. At the end of the study, this loss was statistically significant for the one surviving male in the 30-mg/kg/day group (2.30 kg vs 3.78 kg for the control).

Hematology values at the end of the 1 and 3 months of treatment were similar for the control and the 3 and 10 mg/kg/day groups. At 30 mg/kg/day, the surviving male had decreased numbers of erythrocytes, decreased hemoglobin, decreased hematocrit, and increased platelets. Prothrombin time and activated prothrombin time were also increased. These increases were apparent at 1 month but were much more marked at three months.

Following one month of treatment, glucose was significantly elevated in the 3-mg/kg/day group (117 vs 89 mg/100 ml in the control). The authors of the report attribute this to a single high value for male #7366 who had a value of 131. The other three monkeys in the 3-mg/kg/day groups had levels of 112, 105, and 120-mg/100 ml. Glucose levels in the 10 and 30 mg/kg/day groups were 104 and 122-mg/100 ml, respectively, after one month of treatment. At three months of treatment, glucose levels were 81, 96, 88, and 66-mg/100 ml in the control, 3, 10 and 30 mg/kg/day groups respectively.

There was a decrease in alkaline phosphatase levels in the 30-mg/kg/day group (365 vs 597 IU/l in the control) at one month, which persisted in the one surviving male (360 vs 851 IU/l in the control) at 3 months. Alkaline phosphatase levels in the 3- and 10 mg/kg/day groups at three months were 783 and 743 IU/l showing a dose-related trend toward decreased levels.

SGOT levels were reduced in the 30-mg/kg/day groups at one month (59 vs 29 IU/l in the control) and in the one surviving male at 3 months (88 vs 45 IU/l in the control). SGPT was elevated in both the 10 and 30 mg/kg/day dose groups at 1 month; the levels were 15, 34, and 44 IU/l in the control, 10 and 30 mg/kg/day groups, respectively. SGOT levels in the 10-mg/kg/day

group were comparable to the controls at 3 months (34 vs 31 IU/l in the control) but were still elevated in the one surviving male in the 30-mg/kg/day dose group (46 IU/l).

Cholesterol in the one surviving male in the 30 mg/kg/day group was elevated (240 vs 165 mg/100ml) and total protein and albumin in this animal were reduced. Total protein was 5.52 vs a control level of 8.21 g/100 ml and total albumin was 2.00 vs a control level of 4.82 g/100 ml.

There were no treatment related changes in urinalysis studies at any time period studied.

There were no macroscopic lesions noted at gross necropsy of any animals which died during the study or which were sacrificed at the end of the treatment period.

The following changes in absolute and relative organ weight changes were noted: absolute and relative weight of the hearts in females from the 10 mg/kg/day group were decreased; absolute brain weight of females from this same group were also decreased and relative group mean weight of the pituitary in males from the 3 mg/kg/day group was increased. The significance of these weight changes is difficult to assess, as they were not accompanied by morphologic changes.

One male and two females from the 30 mg/kg/day group and all animals from the 100 mg/kg/day group had marked diffuse lipid depletion in the adrenals. All males and females from the 30 and 100 mg/kg/day groups also had slight to moderate hypocellularity of the bone marrow and moderate atrophy of lymphoid follicles in the spleen. One female from the 30-mg/kg/day group and all animals in the 100-mg/kg/day group had moderate atrophy of the lymphoid follicles in the lymph nodes. No other compound related lesions were seen in at the 30 and 100 mg/kg/day groups. No treatment related lesions were seen in the organs of animals from the 3 and 10 mg/kg/day groups.

The levels of PFOA in the serum and liver are presented below in Table 3. Individual values are presented so there are double entries for most dose levels.

Table 3. Levels of PFOA in the Serum and Liver

Dose	Serum (ppm)		Liver (ppm)		Liver total (ug)	
	Males	Females	Males	Females	Males	Females
0	ND	1	0.05	0.07	3	5
3	53	65	3	7	250	350
3	48	50	ND	ND	ND	ND
10	45	79	9	ND	600	ND
10	71	71	ND	10	ND	750
30	ND	ND	125	80	8000	7500
30	145	ND	60	125	4000	9000
100	ND	ND	100	325	6000	20000

In the first rat study, Goldenthal (1978a) administered CD rats (5/sex/group) dietary levels of 0, 10, 30, 100, 300, and 1000 ppm perfluorooctanoic acid. These dose levels are approximately equivalent to 0.6, 1.7, 5.6, 17.9, and 63.5 mg/kg/day in males, and 0.7, 2.3, 7.7, 22.4 and 76.5 mg/kg/day in females. Animals were housed individually in wire mesh cages and had free access to food and water. Animals were observed twice daily for signs of toxicity and for mortality. Detailed examinations were performed once a week. Individual body weight and food consumption were recorded weekly during the pretest and treatment periods. Blood and urine samples were collected during the pretest period and at 1 and 3 months of the study for hematology and clinical chemistry and urinalysis. At week 13, the serum samples were frozen and shipped to the sponsor for analysis. Organs and tissues from animals that were sacrificed at the end of the study and from two females that died during the treatment period were weighed, examined for gross pathology and samples taken for histopathology. Histopathology was performed on the following organs from rats from the control, 100, 300, and 1000 ppm dose groups: brain with cervical cord, lumbar spinal cord, peripheral nerve, eyes, pituitary, thyroid with parathyroid, adrenals, lung, heart with coronary vessels, aorta, spleen, mesenteric lymph node, thymus, bone with marrow (sternum), salivary gland, small intestines (duodenum, jejunum, ileum) colon, pancreas, liver, kidneys, urinary bladder, testes, ovaries, prostate, uterus, skin (mammary gland), any tissue(s) with gross lesions. Livers from rats from the 10 and 30-ppm dose groups were also examined microscopically and liver samples from all dose groups were frozen and sent to the sponsor for analysis.

One female in the 100 and one female in the 300-ppm group died during collection of blood. These deaths were not considered to be treatment related. All other animals survived until scheduled sacrifice.

There was a significant reduction in mean body weight in males in the 1000-ppm group (362 g vs 466 g in the control group). Food consumption was reduced in males in the 100, 300 and 1000-ppm groups, but the differences were not statistically significant.

Males in the 30, 100, 300 and 1000-ppm groups had significantly reduced numbers of erythrocytes at the end of the treatment period. The values were 7.95, 7.05, 7.16, 6.72, and 6.94 in the control, 30, 100, 300 and 1000-ppm groups, respectively. Males had reduced leukocyte values compared to the controls in all dose groups, but were statistically significant at the 300 ppm group only; leukocyte values were 10.64, 8.88, 9.33, 9.35, 7.63, and 8.06 in the control, 10, 30, 100, 300 and 1000 ppm groups, respectively. A similar phenomenon was seen with hemoglobin values, which were reduced at all dose levels but were significant at the 10-ppm dose level only. Hemoglobin values were 16.2, 14.7, 15.0, 15.4, 14.9, 13.1 in the control, 10, 30, 100, 300 and 1000 ppm groups, respectively. There was no similar effect upon the hematological parameters of female rats in the study.

Males at the 30, 100, 300, and 1000-ppm dose levels had increased glucose levels (mg/100 ml), which were statistically significant at all but the 100-ppm dose level. Reported glucose levels were 121, 120, 136, 134, 143 and 135 mg/100 ml for the 0, 10, 30 100, 300 and 1000 ppm

groups, respectively. B.U.N. levels were elevated in males at the 100, 300, and 1000 ppm dose levels; mean values at 90 days were 20.4, 23.9 and 35.1 mg/100 ml for the three dose groups, respectively, compared to 16.2 mg/100 ml for the controls. Alkaline phosphatase was elevated in males in the 100, 300, and 1000-ppm groups; the levels were 147, 204 and 212 IU/l for the three groups, respectively, compared to 104 IU/l for the controls. Females showed no similar changes in biochemical measurements.

Neither males nor females showed any treatment related changes in urinalysis parameters although females from all groups showed a higher frequency of occult blood in the urine than did males.

The only gross necropsy observation was noted in males at the 1000-ppm dose level. These animals had enlarged livers that showed varying degrees of surface discoloration. Neither females from the 1000-ppm dose level nor males or females from the lower dose levels showed such effects.

Both absolute and relative liver weights were significantly increased in males in the 30, 300 and 1000-ppm groups and in one female in the 1000-ppm group. Compound-related liver lesions occurred in all male rats in the 100, 300 and 1000-ppm groups. These lesions consisted of focal to multifocal, very slight-to-slight hypertrophy of hepatocytes in centrilobular to midzonal regions of the affected liver lobules. In some instances these lesions were accompanied by an increased amount of yellowish-brown pigment resembling lipofuscin in the cytoplasm of hepatocytes and occasionally in sinusoidal lining cells. The incidence and severity of the lesions was more pronounced among male rats at the 1000-ppm dietary level.

A comparison of the serum levels of PFOA is shown below in Table 4. The greater toxicity observed in the males than in the females is due to the gender difference in elimination as demonstrated by the differences in serum PFOA levels.

Table 4. Comparison of Male and Female PFOA Serum Levels

Dose	PFOA in Serum (ppm)	
	Males	Females
0	0	0
10	21	ND
30	34	0.15
100	36	ND
300	38	0.25
1000	49	0.65

ND = Not Determined.

In the second rat study, Palazzolo (1993) administered 45-55 male Sprague-Dawley rats per group, doses of 1, 10, 30, or 100 ppm (approximate mean compound consumption at week 13 of

0.05, 0.47, 1.44, and 4.97 mg/kg/day) APFO *ad libitum* in the diet for 13 weeks. Two control groups (a nonpair-fed control group and a control group pair-fed to the 100 ppm dose group) were also exposed during that period. Following the 13-week exposure period, 10 animals per group were fed basal diet for an additional 8-weeks post-treatment and observed for any signs of recovery. All test diets were assayed and evaluated for test material homogeneity and stability. All animals were observed twice daily for mortality, moribundity, and general clinical signs of toxicity. Body weights were recorded once before exposures began, weekly during the treatment period, and then on the day of necropsy. Food consumption was recorded weekly for all groups, including the nonpair-fed control group; daily for the pair-fed animals, and then weekly for all of the animals retained for the recovery phase of the study. A total of 15 animals per group were sacrificed following 4, 7, or 13 weeks of treatment; 10 animals per group were sacrificed after 13 weeks of treatment and following 8 weeks of non-treatment. Serum samples collected from 10 animals per group at each scheduled sacrifice during treatment and from 5 animals per group during recovery were analyzed for estradiol, total testosterone, luteinizing hormones, and for test material residue. The level of palmitoyl CoA oxidase, an indicator of peroxisome proliferation, was analyzed from a section of liver that was obtained from 5 animals per group at each scheduled sacrifice. The following organs from all animals at each scheduled sacrifice were weighed: brain, liver, lungs, testis (one), seminal vesicle, prostate, coagulating gland, and urethra. The following tissues in these same animals were preserved in 10% phosphate-buffered formalin and examined macroscopically: external surface of the body, all orifices, the cranial cavity, the external surfaces of the brain and spinal cord, the nasal cavity and paranasal sinuses; the thoracic, abdominal, and pelvic cavities and viscera; and also examined microscopically: any observed lesions, brain, liver, lungs, testes (one), seminal vesicle, prostate, coagulating gland, and urethra. In addition, the following tissues were preserved in glutaraldehyde for electron microscopic examination: brain, liver, lungs, testes (one), seminal vesicle, and prostate.

In the analysis of the data, animals in groups exposed to 1, 10, 30, and 100 ppm APFO were compared to the control animals in the nonpair-fed group, while the data from the pair-fed control animals were compared to animals exposed to100 ppm APFO. All test diets were considered to be homogeneous and stable under the experimental conditions. All animals survived to scheduled sacrifice, with the exception of one animal in the 100-ppm dosed-group that was sacrificed on week 4 due to severe neck sores unrelated to treatment. Twice-daily examinations of all animals were unremarkable. At 100 ppm, significant reductions in body weights were seen compared to the pair-fed control group during week 1 and the nonpair-fed control group during weeks 1-13 (i.e., throughout treatment). During recovery, however, no reductions in body weights were apparent. Body weight data in the other dosed-groups were comparable to controls. At 100 ppm, mean body weight gains were significantly higher than the pair-fed control group during week 1 and significantly lower than the nonpair-fed control group during weeks 1-13. At 10 and 30 ppm, mean body weight gains were significantly lower than the nonpair-fed control group at week 2. These differences in body weight gains were not observed during the recovery period. Significant differences in food consumption were observed at 100 ppm during weeks 1 and 2 only, when compared to the nonpair-fed control group; no other significant differences in food consumption were noted. There were no significant differences among the groups for any of the hormones evaluated in the serum. Likewise, serum

analysis of test material residue showed no increase in serum APFO levels over the course of treatment. Statistically significant higher hepatic palmitoyl CoA oxidase activity was observed at 30 and 100 ppm; however, this effect returned to control levels by the end of the recovery period. At 10 ppm, statistically significant higher levels of hepatic palmitoyl CoA oxidase activity were observed at week 5 only. Mean enzyme activities were highest during week 8 for animals exposed to 10, 30, and 100 ppm. Significant increases in absolute and relative liver weights and hepatocellular hypertrophy were observed at weeks 4, 7, or 13 in the 10, 30 and 1000 ppm groups. The authors suggested that these changes might be associated with peroxisome proliferation, especially since increases in hepatic palmitoyl CoA oxidase activity were also observed at this dose level during treatment. During recovery, however, none of the liver effects were observed, indicating that these treatment-related liver effects were reversible.

Therefore, under the conditions of this study, a NOAEL of 1.0 ppm (0.05 mg/kg/day) and a LOAEL of 10 ppm (0.47 mg/kg/day) are indicated based on reductions in body weight and body weight gain, and on increases in absolute and relative liver weights with hepatocellular hypertrophy.

Thomford (2001a,b) recently conducted a range-finding and a 6-month toxicity study in cynomolgus monkeys. In the range-finding study, Thomford (2001a) administered male cynomolgus monkeys an oral capsule containing 0, 2, or 20 mg/kg/day APFO for 4 weeks. There were 3 monkeys in the 2 and 20 mg/kg/day groups and one monkey in the control group. The monkeys weighed 2.1 to 3.6 kg at the start of treatment. Animals were observed twice daily for mortality and moribundity and were examined at least once daily for signs of poor health or abnormal behavior. Food consumption was assessed qualitatively. Body weight data were recorded before the start of treatment, on day 1 of treatment and weekly thereafter. The monkeys were fasted overnight and blood samples were collected one week prior to the start of the study and on day 30 for clinical hematology and clinical chemistry analyses, and hormone and PFOA level. Blood for clinical chemistry was also collected from each animal on day 2 (approximately 24 hours after the first dose). Samples were analyzed for estradiol, estrone, estriol, thyroid stimulating hormone, total and free triiodothyronine, and total and free thyroxin.

At scheduled necropsy, samples of the right lateral lobe of the liver were collected from each animal and analyzed for palmitoyl CoA oxidase activity. Representative samples of liver, right and left testes, and pancreas were collected from each animal for cell proliferation evaluation using proliferation cell nuclear antigen. Bile was collected from each animal for bile acid determination. A sample of liver was collected from each animal for PFOA concentration analysis. The following tissues (when present) or representative samples were collected and preserved in 10 % neutral-buffered formalin, unless otherwise specified, for possible future examination: adrenal (2), aorta, brain, cecum, colon, duodenum, epididymis (2), esophagus, eyes [preserved in Davidson's fixative (2)], femur with one marrow (articular surface of the distal end), gallbladder, heart, ileum, jejunum, kidney (2), lesions, liver, lung, lymph node (mesenteric) mammary gland, muscle (thigh), pancreas, pituitary, prostate, rectum, salivary gland [mandibular (2)], sciatic nerve, seminal vesicle (2) skin, spinal cord (cervical, thoracic, and lumbar), spleen, sternum with bone marrow, stomach, testis (2), thymus, thyroid (2) with parathyroid, trachea,

and urinary bladder. The adrenals, liver, pancreas, spleen, and testes from each animal were embedded in paraffin, sectioned, stained with hematoxylin and eosin and examined microscopically.

All animals survived to scheduled sacrifice. There were no clinical signs of toxicity in the treated groups and there was no effect on body weight. Low or no food consumption was observed for one animal given 20 mg/kg/day. No food consumption was noted for this animal on day 12 and low food consumption was noted on days 5, 7, 11, 14 17, and 24, respectively. For this animal, decreased food consumption is in all likelihood related to APFO administration. There were no effects on estradiol, estriol, thyroid stimulating hormone, total and free triiodothyronine, and total and free thyroxin. Estrone levels were notably lower for males given 2 and 20 mg/kg/day APFO. There was no evidence of peroxisome proliferation or cell proliferation in the liver, testes or pancreas of treated monkeys. No adverse effects were noted in either gross or clinical pathology studies.

In the 26-week study, male cynomolgus monkeys were administered APFO by oral capsule at doses of 0, 3, 10 or 30 mg/kg/day for 26 weeks (Thomford, 2001b; Butenhoff et al., 2002). At study initiation the monkeys weighed 3.2 to 4.5 kg. There were 4 monkeys in the 3 mg/kg/day group and 6 monkeys in each of the other groups. Dosing of animals in the 30 mg/kg/day dose group was stopped from days 11–21 because of toxicity. When dosing was resumed on day 22, animals received 20 mg/kg/day and this group was designated the 30/20 mg/kg/day group. At the end of the 26-week treatment period, 2 animals in the control and 10 mg/kg/day groups were observed for a 13-week recovery period.

Animals were observed twice daily for mortality and moribundity and were examined at least once daily for signs of poor health or abnormal behavior; food consumption was assessed qualitatively. Ophthalmic examinations were done before initiation of treatment and during weeks 26 and 40. Body weight data were recorded weekly before the start of treatment, on day 1 of treatment and weekly thereafter. Blood and urine samples were collected for clinical hematology, clinical chemistry, and urinalysis before the start of treatment and on days 11, 31, 63, 91, 182, 217, 245 and 275. Blood samples were also taken for hormone determinations; samples were analyzed for estradiol, estrone, estriol, thyroid stimulating hormone, total and free triiodothyronine, total and free thyroxin, and testosterone. Blood, urine and feces were collected during week 2 and every 2 weeks thereafter during treatment and recovery for PFOA concentration analyses. The animals were not fasted before collections.

At scheduled necropsy, liver samples were taken for determination of PFOA levels. The right lateral lobe of the liver was collected from each animal for palmitoyl CoA oxidase activity analyses, and representative samples of liver, right and left testes, and pancreas were collected from each animal for cell proliferation evaluation using proliferation cell nuclear antigen. All available bile was collected for bile acid determination. The following organs were weighed at scheduled and unscheduled sacrifices; paired organs were weighed separately: adrenal (2), brain, epididymis (2), kidney (2), liver, pancreas, testis (2), and thyroid (2) with parathyroid. Organ to body weight percentages and organ to brain weight ratios were calculated. The following tissues

were collected for histopathology: adrenal (2), aorta, brain, cecum, colon, duodenum, epididymis (2), esophagus, eyes [preserved in Davidson's fixative (2)], femur with bone marrow (articular surface of the distal end), gallbladder, heart, ileum, jejunum, kidneys (2), lesions, liver, lung, mesenteric lymph node, mammary gland, pancreas, pituitary, prostate, rectum, salivary gland [mandibular (2)], sciatic nerve, seminal vesicle (2), skeletal muscle (thigh), skin, spinal cord (cervical, thoracic, and lumbar), spleen, sternum with bone marrow, stomach, testis [(2) preserved in Bouin's solution, thymus, thyroid (2) with parathyroid, trachea and urinary bladder.

Two animals, one male from the 30/20 mg/kg/day dose group and one male from the 3 mg/kg/day dose group, were sacrificed in moribund condition during the study. The male in the 30/20 mg/kg/day dose group was sacrificed on day 29. This animal exhibited signs of hypoactivity, weight loss, few or no feces, low or no food consumption and the entire body was cold to the touch before death. Necropsy revealed esophageal and gastric lesions that were indicative of an injury that occurred during dosing and liver lesions that were presumed to be treatment related. The animal from the 3 mg/kg/day dose group was sacrificed on day 137. This animal showed clinical signs of limited use and paralysis of the hind limbs, ataxia and hypoactive behavior, few feces and no food consumption. The cause of death was not determined, but APFO treatment could not be ruled out.

Males given 30 mg/kg/day from days 1-11 had clinical signs of few feces and low food consumption and they lost weight during week 1 of treatment. Based on these signs, treatment was stopped on day 11 and was not resumed until day 22. When treatment was resumed, the dose was lowered to 20 mg/kg/day; this group was then designated the 30/20 mg/kg/day group. Of the remaining animals in this group, only 2 tolerated this dose level for the remaining 23 weeks of treatment. Treatment of three males given 30/20 mg/kg/day was halted on days 43 (week 7), 66 (week 10), and 81 (week 12) respectively. Clinical signs in these animals included thin appearance, few or no feces, low or no food consumption, and weight loss. The animals appeared to recover from compound-related effects within 3 weeks after cessation of treatment.

Mean body weight changes were notably lower during weeks 1 and 2 for males receiving 30 mg/kg/day. During week 2, this change was statistically significant. Treatment was stopped on day 11 and when it was resumed at 20 mg/kg/day on day 21, mean body weight changes were significantly lower than controls during weeks 7, 9 and 24. Overall mean body weight changes through week 27 were notably lower for the males in the 30/20 mg/kg/day group. There was an increased incidence of low or no food consumption for animals in the 30/20 mg/kg/day group that was considered to be treatment related.

There were no consistent or clearly dose-related effects on estrone, estradiol, estriol, thyroid stimulating hormone, or testosterone that were seen in any treatment group over time. In general, thyroid hormones were decreased beginning on day 35 in animals in the 10 or 30/20 mg/kg/day groups. Triiodothyronine remained depressed through day 183 in the 30/20 mg/kg group. Total thyroxin was decreased beginning on day 35 in animals administered 10 or 30/20 mg/kg APFO. The effect on thyroxin was most pronounced at day 35 in animals administered 10 mg/kg/day and day 66 in animals administered 30/20 mg/kg/day. Thereafter, the effect began

to diminish and recovery was observed either in the last 3 months of dosing or during the recovery phase of the experiment. No changes in cholecystokinin concentrations were seen over time in either dose group.

Two males from the 30/20 mg/kg/day group, the one sacrificed on day 29 and one for whom treatment was stopped because of poor health, had moderate to marked increased serum enzyme concentrations (i.e. aspartate aminotransferase, alanine aminotransferase, sorbitol dehydrogenase, and creatine kinase) and a mildly increased serum bile acid concentration. At 30/20 mg/kg/day, there were slight increases in triglyceride concentrations and mild to moderate decreases in absolute neutrophil count, total protein concentration and albumin concentration. Two animals in the 30/20 mg/kg/day group had markedly increased serum enzyme activities and mildly increased serum bile acid concentrations. At 3 mg/kg/day or 10 mg/kg/day, APFO had no effects on hematology, coagulation, clinical chemistry, or urinalysis. There was no evidence of persistent or delayed toxic effects on clinical pathology test results during the recovery period.

At terminal sacrifice at 26 weeks, there were statistically significant increases in absolute and relative liver weights in all dose groups. The absolute liver weights were increased by 35, 38 and 50% in the 3, 10 and 30/20 mg/kg/day groups, respectively; relative liver weights were increased by 20, 20 and 60% in the 3, 10 and 30/20 mg/kg/day groups, respectively. In addition, mean liver-to-brain weight was significantly increased in the 10 mg/kg/day group. The increased liver weights were considered treatment related. There were no treatment-related macroscopic or microscopic changes in any organs at the terminal sacrifice, including liver, adrenal, spleen, pancreas, and testis. While there was no evidence of enhanced cell proliferation in the testes or pancreas of the treated monkeys, the findings in the liver were equivocal. There was evidence of mitochondrial proliferation in the livers of treated monkeys (Butenhoff et al., 2002).

At the recovery sacrifice, there were no treatment-related effects on terminal body weights or on absolute or relative organ weights at recovery sacrifice indicating that the liver weight changes seen at terminal sacrifice were reversible over time. There were no treatment-related macroscopic or microscopic changes at the recovery sacrifice.

The results of the analyses of PFOA in the serum, liver, urine and feces are presented in Tables 5-8, respectively (3M Environmental Laboratory, 2001c). Low levels of PFOA were often detected in the sera, liver, urine and feces of the control animals. The levels in treated animals were significantly higher than those seen in the controls. In general, PFOA levels in the sera of test animals increased with dose but decreased over time during treatment. On day 9 of treatment, the serum levels were 126 ± 36.1 µg/mL in the 3 mg/kg/day group and 1597 ± 2392 µg/mL in the 30/20 mg/kg/day group, and during weeks 26/27, the serum levels were 52.5 ± 9.14 µg/mL in the 3 mg/kg/day group and 51.5 ± 77.6 µg/mL in the 30/20 mg/kg/day group. PFOA levels in serum decreased over time during recovery until they reached 1.18 ± 0.827 µg/mL in the 10 mg/kg/day group as compared to 0.0738 ± 0.00256 in the controls during week 40.

A similar trend was seen in the urine. On day 9, the levels in the urine were 73.5 ± 38.1, 221 ±

124, and 909 ± 209 µg/mL in the 3, 10 and 30/20 mg/kg/day groups, respectively, and during week 26, the levels were 51.6 ± 13.7, 109 ± 75.2, and 19.2 ± 27.0 µg/g, respectively, in these same groups. During week 2, the levels in the feces were less than the limit of quantitation in the control animals, and 7.43 ± 6.54, 15.4 ± 10.2 and 56.6 ± 73.7 ug/g in the 3, 10, and 30/20 mg/kg/day groups, respectively; during week 26 the levels were 2.92 ±1.35, 43.0 ± 36.9 and 10.3 ± 20.8 µg/g in the 3, 10, and 30/20 mg/kg/day groups, respectively. There is no explanation for the high levels of PFOA seen in the feces of the control animals during week 22. During the recovery period, PFOA levels in both urine and feces fell to levels that were comparable to control levels.

Under the conditions of the study, the LOAEL was 3 mg/kg/day (liver toxicity and possibly mortality) and a NOAEL was not established.

Table 5
Average PFOA Concentrations in the Serum (μg/mL) of Treated Animals

	0 mg/kg/day	**3 mg/kg/day**	**10 mg/kg/day**	**30/20 mg/kg/day**
Time Point				
Day 9	0.0613 ± 0.0472	126 ± 36.1	189 ± 48.9	1597 ± 2392
Week 4	0.0206 ±0.0105	98.4 ± 42.7	172 ± 71.9	1084 ±1839
Week 6	0.103 ± 0.0113	102 ± 33.6	95.8 ± 20.6	145 ± 21.6
Week 8	<LOQ	94.7 ± 27.3	97.6 ± 23.5	166 ± 98.9
Week 10	0.126 ± 0.0348	105 ± 37.7	93.6 ± 13.7	237 ± 158
Week 12	0.123 ± 0.0507	79.6 ± 25.9	90.6 ± 24.5	140 ± 77.5
Week 14	0.162 ± 0.0643	90.0 ± 28.9	92.2 ± 27.6	79.4 ± 28.3
Week 16	0.128 ± 0.0721	68.6 ± 26.0	98.5 ± 42.3	83.9 ± 58.5
Week 18	0.183 ± 0.0637	29.4 ± 25.4	17.4 ± 14.3	36.2 ± 21.2
Week 20	0.224 ± 0.0730	33.0 ± 26.0	96.6 ± 26.6	97.7 ± 129
Week 22	0.232 ± 0.131	77.6 ± 24.9	105 ± 36.3	58.6 ± 45.6
Week 24	<LOQ	72.2 ± 68.8	90.9 ± 21.2	62.7 ± 74.3
Week 26	0.209 ± 0.156	118 ± 27.5	77.4 ± 16.9	77.8 ± 126
Week 26/27	0.223 ± 0.105	52.5 ± 9.14	74.1 ± 33.1	51.5 ± 77.6
Week 28	0.181 ± 0.0391	NS	25.9 ± 7.07	NS
Week 30	0.144 ± 0.0238	NS	11.3 ± 5.11	NS
Week 32	0.110 ± 0.0216	NS	7.60 ± 3.90	NS
Week 34	0.0861 ± 0.0256	NS	3.97 ± 2.09	NS
Week 36	0.111 ± 0.0445	NS	2.75 ± 1.88	NS
Week 38	0.0941 ± 0.0324	NS	1.84 ± 1.40	NS
Week 40	0.0738 ± 0.00256	NS	1.18 ± 0.827	NS

LOQ = Limit of Quantitation; NS = No Sample; Results are expressed as group averages ± the standard deviation associated with that group. Data are accurate to within one SD of the average fortified sample recovery. The average fortified sample recovery was 94% with an SD of 11%.

Table 6
Average PFOA Concentrations in the Liver (μg/g) of Treated Animals

	0 mg/kg/day	**3 mg/kg/day**	**10 mg/kg/day**	**30/20 mg/kg/day**
Time Point				
Week 20	NS	18.3	NS	NS
Week 27	0.117 ± 0.0730	15.3 ± 3.02	14.0 ± 7.55	42.8 ± 63.3
Week 40	<LOQ	NS	0.114 ± 0.0441	NS

LOQ = Limit of Quantitation; NS = No Sample
Results are expressed as group averages ± the standard deviation associated with that group. Data are accurate to within one SD of the average fortified sample recovery. The average fortified sample recovery was 90% with an SD of 26%.

Table 7. Average PFOA Concentrations in the Urine (µg/mL) of Treated Animals

	0 mg/kg/day	3 mg/kg/day	10 mg/kg/day	30/20 mg/kg/day
Time Point				
Week 2	< LOQ	73.5 ± 38.1	221 ± 124	909 ± 269
Week 4	0.152 ± 0.337	54.9 ± 4.62	190 ± 91.6	240 ± 161
Week 6	0.0587 ± 0.0716	65.7 ± 46.9	128 ± 50.0	272 ± 140
Week 8	0.0161 ± 0.00940	47.6 ± 20.6	206 ± 73.1	180 ± 109
Week 10	0.0177 ± 0.00114	39.9 ± 18.7	175 ± 92.3	359 ± 449
Week 12	0.0141 ± 0.00648	48.8 ± 18.8	201 ± 92.7	118 ± 111
Week 14	7.96 ± 19.5	63.1 ± 47.2	139 ± 52.5	72.9 ± 84.0
Week 16	0.0299 ± 0.0339	50.2 ± 21.0	139 ± 57.0	50.2 ± 67.9
Week 18	0.0256 ± 0.0248	37.7 ± 19.3	186 ± 63.9	43.1 84.6
Week 20	0.0211 ± 0.0130	52.1 ± 9.63	144 ± 135	44.0 ± 59.9
Week 22	0.0231 ± 0.00688	95.8 ± 80.8	158 ± 78.4	98.5 ± 134
Week 24	0.0125 ± 0.00749	46.3 ± 8.52	157 ± 63.3	56.0 ± 77.2
Week 26	0.0268 ± 00.0265	51.6 ± 13.7	109 ± 75.2	19.2 ± 27.0
Week 28	0.118 ± 0.142	NS	0.327 ± 0.0182	NS
Week 30	<LOQ	NS	0.361 ± 0.118	NS
Week 32	<LOQ	NS	0.114 ± 0.0608	NS
Week 34	<LOQ	NS	0.121 ± 0.0305	NS
Week 36	0.0117 ± 0.00841	NS	0.0502 ± 0.0166	NS
Week 38	<LOQ	NS	0.0284 ± 0.0102	NS
Week 40	<LOQ	NS	0.0254 ± 0.0101	NS

LOQ = Limit of Quantitation; NS = No Sample; Results are expressed as group averages ± the standard deviation associated with that group. Data are accurate to within one SD of the average fortified sample recovery. The average fortified sample recovery was 88% with an SD of 17%.

Table 8
Average PFOA Concentrations in the Feces (μg/g) of Treated Animals

	0 mg/kg/day	**3 mg/kg/day**	**10 mg/kg/day**	**30/20 mg/kg/day**
Time Point				
Week 2	<LOQ	7.43 ± 6.54	15.4 ± 10.2	56.6 ± 73.7
Week 4	0.0214 ± 0.0178	10.4 ± 12.0	23.4 ± 10.6	22.0 ± 6.23
Week 6	0.108 ± 0.00192	12.1 ± 14.1	23.3 ± 8.46	101 ± 86.7
Week 8	0.0782 ± 0.103	9.46 ± 9.21	41.0 ± 25.0	36.7 ± 34.2
Week 10	<LOQ	3.96 ± 3.68	26.0 ± 17.4	48.0 ± 34.0
Week 12	0.0498 ± 0.0894	7.15 ± 5.65	10.3 ± 6.07	32.0 ± 42.9
Week 14	0.139 ± 0.308	7.50 ± 2.43	27.2 ± 29.4	19.2 ± 25.2
Week 16	0.0572 ± 0.0762	6.88 ± 2.62	31.4 ± 23.3	18.2 ± 28.8
Week 18	0.258 ± 0.654	5.72 ± 7.15	17.3 ± 13.3	22.1 ± 31.7
Week 20	0.405 ± 1.08	6.81 ± 4.89	52.4 ± 39.5	37.8 ± 58.1
Week 22	15.5 ± 36.9	13.8 ± 5.22	39.5 ± 21.0	25.2 ± 36.0
Week 24	0.517 ± 1.13	6.22 ± 5.45	40.5 ± 21.8	34.6 ± 47.7
Week 26	0.0172 ± 0.00892	2.92 ± 1.35	43.0 ± 36.9	10.3 ± 20.8
Weeks 28-34	0.279 ± 0.732	NS	0.387 ± 0.372	NS
Weeks 36-40	0.0103 ± 0.00684	NS	0.0336 ± 0.0313	NS

LOQ = Limit of Quantitation; NS = No Sample
Results are expressed as group averages ± the standard deviation associated with that group. Data are accurate to within one SD of the average fortified sample recovery. The average fortified sample recovery was 117% with an SD of 22%.

3.6 Developmental Toxicity Studies in Animals

Three prenatal developmental toxicity studies of APFO have been conducted, one inhalation and two oral studies.

The first of these studies was an oral developmental toxicity study in rats (Gortner, 1981). Based on the results of a range-finding study, an upper dose level of 150 mg/kg/day was set for the definitive study in which five groups of 22 time-mated Sprague-Dawley rats were administered 0, 0.05, 1.5, 5, and 150 mg/kg/day APFO in distilled water by gavage on gestation days (GD) 6-15. Doses were adjusted according to body weight. Dams were monitored on GD 3-20 for clinical signs of toxicity. Individual body weights were recorded on GD 3, 6, 9, 12, 15, and 20.

Animals were sacrificed on GD 20 by cervical dislocation and the ovaries, uteri, and contents were examined for the number of corpora lutea, number of viable and non-viable fetuses, number of resorption sites, and number of implantation sites. Fetuses were weighed and sexed and subjected to external gross necropsy. Approximately one-third of the fetuses were fixed in Bouin's solution and examined for visceral abnormalities by free-hand sectioning. The remaining fetuses were subjected to skeletal examination using alizarin red.

Signs of maternal toxicity consisted of statistically significant reductions in mean maternal body weights on GD 9, 12, and 15 at the high-dose group of 150 mg/kg/day. Mean maternal body weight on GD 20 continued to remain lower than controls, although the difference was not statistically significant. Other signs of maternal toxicity that occurred only at the high-dose group included ataxia and death in three rat dams. No other effects were reported. Administration of APFO during gestation did not appear to affect the ovaries or reproductive tract of the dams. Under the conditions of the study, a NOAEL of 5 mg/kg/day and a LOAEL of 150 mg/kg/day for maternal toxicity were indicated.

A significantly higher incidence in fetuses with one missing sternebrae was observed at the high-dose group of 150 mg/kg/day; however this skeletal variation also occurred in the controls and the other three dose groups (at similar incidence but lower than the high-dose group) and therefore was not considered to be treatment-related. No significant differences between treated and control groups were noted for other developmental parameters that included the mean number of males and females, total and dead fetuses, the mean number of resorption sites, implantation sites, corpora lutea and mean fetus weights. Likewise, a fetal lens finding initially described as a variety of abnormal morphological changes localized to the area of the embryonal nucleus, was later determined to be an artifact of the free-hand sectioning technique and therefore not considered to be treatment-related. Under the conditions of the study, a NOAEL for developmental toxicity of 150 mg/kg/day (highest dose group) was indicated.

A second oral prenatal developmental toxicity study was conducted in rabbits (Gortner, 1982). Based on the results of a range-finding study, an upper dose level of 50 mg/kg/day was set for the definitive study in which four groups of 18 pregnant New Zealand White rabbits were administered 0, 1.5, 5, and 50 mg/kg/day APFO in distilled water by gavage on gestation days (GD) 6-18. Pregnancy was established in each sexually mature female by i.v. injection of pituitary lutenizing hormone in order to induce ovulation, followed by artificial insemination with 0.5 ml of pooled semen collected from male rabbits; the day of insemination was designated as day 0 of gestation. A constant dose volume of 1 ml/kg was administered. Individual body weights were measured on GD 3, 6, 9, 12, 15, 18, and 29. The does were observed daily on GD 3-29 for abnormal clinical signs. On GD 29, the does were euthanized and the ovaries, uterus and contents examined for the number of corpora lutea, live and dead fetuses, resorptions and implantation sites. Fetuses were examined for gross abnormalities and placed in a 37^0 C incubator for a 24-hour survival check. Pups were subsequently euthanized and examined for visceral and skeletal abnormalities. A blood sample was taken from six does prior to dosing and then on GD 18 and 29; a liver sample was taken from the same animals on GD 29. All samples were sent to the sponsor for analysis. This information was unavailable at the time of this

review.

Signs of maternal toxicity consisted of statistically significant transient reductions in body weight gain on GD 6-9 when compared to controls; body weight gains returned to control levels on GD 12-29. Administration of APFO during gestation did not appear to affect the ovaries or reproductive tract contents of the does. No clinical or other treatment-related signs were reported. Under the conditions of the study, a NOAEL of 50 mg/kg/day, the highest dose tested, for maternal toxicity was indicated.

No significant differences were noted between controls and treated groups for the number of males and females, dead or live fetuses, and fetal weights. Likewise, there were no significant differences reported for the number of resorption and implantation sites, corpora lutea, the conception incidence, abortion rate, or the 24-hour mortality incidence of the fetuses. Gross necropsy and skeletal/visceral examinations were unremarkable. The only sign of developmental toxicity consisted of a dose-related increase in a skeletal variation, extra ribs or 13^{th} rib, with statistical significance at the high-dose group (38% at 50 mg/kg/day, 30% at 5 mg/kg/day, 20% at 1.5 mg/kg/day, and 16 % at 0 mg/kg/day). A statistically significant increase in 13^{th} ribs-spurred occurred in the mid-dose group of 5 mg/kg/day; however, the biological significance of this effect is uncertain since in both the high- and low-dose groups, this effect occurred at the same rate and was not statistically significantly different from controls. Therefore, under the conditions of the study, a LOAEL for developmental toxicity of 50 mg/kg/day (highest dose group) was indicated.

Staples et al. (1984) also conducted a developmental toxicity study of APFO. The study design consisted of an inhalation and an oral portion, each with two trials or experiments. The first trial was the teratology portion of the study, in which the dams were sacrificed on GD 21; while in the second trial, the dams were allowed to litter and the pups were sacrificed on day 35-post partum. For the inhalation portion of the study, the two trials consisted of 12 pregnant Sprague-Dawley rats per group exposed to 0, 0.1, 1, 10, and 25 mg/m^3 APFO for 6 hours/day, on GD 6-15. In the oral portion of the study, 25 and 12 Sprague-Dawley rats for the first and second trials, respectively, were administered 0 and 100 mg/kg/day APFO in corn oil by gavage on GD 6-15. For both routes of administration, females were mated on an as-needed basis and when the number of mated females was bred, they were ranked within breeding days by body weight and assigned to groups by rotation in order of rank. Finally, two additional groups (six dams per group) were added to each trial that was pair-fed to the 10 and 25 mg/m^3 groups.

For the teratology portion of the study (trial one), dams were weighed on GD 1, 6, 9, 13, 16, and 21 and observed daily for abnormal clinical signs. On GD 21, the dams were sacrificed by cervical dislocation and examined for any gross abnormalities, liver weights were recorded and the reproductive status of each animal was evaluated. The ovaries, uterus and contents were examined for the number of corpora lutea, live and dead fetuses, resorptions and implantation sites. Pups (live and dead) were counted, weighed and sexed and examined for external, visceral, and skeletal alterations. The heads of all control and high-dosed group fetuses were examined for visceral alterations as well as macro- and microscopic evaluation of the eyes.

For trial two, in which the dams were allowed to litter, the procedure was the same as that for trial one up to GD 21. Two days before the expected day of parturition, each dam was housed in an individual cage. The date of parturition was noted and designated Day 1 PP. Dams were weighed and examined for clinical signs on Days 1, 7, 14, and 22 PP. On Day 23 PP all dams were sacrificed. Pups were counted, weighed, and examined for external alterations. Each pup was subsequently weighed and inspected for adverse clinical signs on Days 4, 7, 14, and 22 PP. The eyes of the pups were also examined on Days 15 and 17 PP for the inhalation portion and on Days 27 and 31 PP for the gavage portion of the study. Pups were sacrificed on Day 35 PP and examined for visceral and skeletal alterations.

Inhalation Exposure

Trial One:

Treatment-related clinical signs of maternal toxicity for trial one (teratology) occurred at 10 and 25 mg/m^3 and consisted of wet abdomens, chromodacryorrhea, chromorhinorrhea, a general unkempt appearance, and lethargy in four dams at the end of the exposure period (high-concentration group only). Three out of 12 dams died during treatment at 25 mg/m^3 (on GD 12, 13, and 17). Food consumption was significantly reduced at both 10 and 25 mg/m^3; however, no significant differences were noted between treated and pair-fed groups. Significant reductions in body weight were also observed at these concentrations, with statistical significance at the high-concentration only. Likewise, statistically significant increases in mean liver weights were seen at the high-concentration group. Under the conditions of the study, a NOAEL and LOAEL for maternal toxicity of 1 and 10 mg/m^3, respectively, were indicated.

No effects were observed on the maintenance of pregnancy or the incidence of resorptions. Mean fetal body weights were significantly decreased in the 25-mg/m^3 groups and in the control group pair-fed 25 mg/m^3. A detailed microscopic visceral and eye examination of the fetuses did not reveal any treatment-related effects; however in the control group that was pair-fed 25 mg/m^3, a statistically significant increased incidence of fetuses with partially ossified sternebrae was observed. Under the conditions of the study, a NOAEL and LOAEL for developmental toxicity of 10 and 25 mg/m^3, respectively, were indicated.

Trial Two:

Clinical signs of maternal toxicity seen at 10 and 25 mg/m^3 were similar in type and incidence to those described for trial one. Maternal body weight gain during treatment at 25 mg/m^3 was less than controls, although the difference was not statistically significant. In addition, 2 out of 12 dams died during treatment at 25 mg/m^3. No other treatment-related effects were reported, nor were any adverse effects noted for any of the measurements of reproductive performance. Under the conditions of the study, a NOAEL and LOAEL for maternal toxicity of 1 and 10 mg/m^3, respectively, were indicated.

Signs of developmental toxicity in this group consisted of statistically significant reductions in pup body weight on Day 1 PP (6.1 g at 25 mg/m^3 vs. 6.8 g in controls). On Days 4 and 22 PP, pup body weights continued to remain lower than controls, although the difference was not statistically significant (Day 4 PP: 9.7 g at 25 mg/m^3 vs.10.3 in controls; Day 22 PP: 49.0 g at 25 mg/m^3 vs. 50.1 in controls). No significant effects were reported following external examination of the pups or with ophthalmoscopic examination of the eyes. Under the conditions of the study, a NOAEL and LOAEL for developmental toxicity of 10 and 25 mg/m^3, respectively, were indicated.

Oral Exposure

Trial One:

Three out of 25 dams died during treatment of 100 mg/kg APFO during gestation (one death on GD 11; two on GD 12). Clinical signs of maternal toxicity in the dams that died were similar to those seen with inhalation exposure. Food consumption and body weights were reduced in treated animals compared to controls. No adverse signs of toxicity were noted for any of the reproductive parameters such as maintenance of pregnancy or incidence of resorptions. Likewise, no significant differences between treated and control groups were noted for fetal weights, or in the incidences of malformations and variations; nor were there any effects noted following microscopic examination of the eyes.

Trial Two:

Similar observations for clinical signs were noted for the dams as in trial one. Likewise, no adverse effects on reproductive performance or in any of the fetal observations were noted.

3.7 Reproductive Toxicity Studies in Animals

York (2002) conducted a two-generation reproductive toxicity study of APFO. Five groups of 30 Sprague-Dawley rats per sex per dose group were administered APFO by gavage at doses of 0, 1, 3, 10, and 30 mg/kg/day six weeks prior to and during mating. Treatment of the F0 male rats continued until mating was confirmed, and treatment of the F0 female rats continued throughout gestation, parturition, and lactation.

The F0 animals were examined twice daily for clinical signs, abortions, premature deliveries, and deaths. Body weights of F0 male rats were recorded weekly during the dosage period and then on the day of sacrifice. Body weights of F0 female rats were recorded weekly during the pre- and cohabitation periods and then on gestation days (GD) 0, 7, 10, 14, 18, 21, and 25 (if necessary) and on lactation days (LD) 1, 5, 8, 11, 15, and 22 (terminal body weight). Food consumption values in F0 male rats were recorded weekly during the treatment period, while in F0 female rats, values were recorded weekly during the precohabitation period, on GDs 0, 7, 10, 14, 18, 21, and 25 and on LDs 1, 5, 8, 11, and 15.

Estrous cycling was evaluated daily by examination of vaginal cytology beginning 21 days before the scheduled cohabitation period and continuing until confirmation of mating by the presence of sperm in a vaginal smear or confirmation of a copulatory plug. On the day of scheduled sacrifice, the stage of the estrous cycle was assessed.

For mating, one male rat and one female rat per group were cohabitated for a maximum of 14 days. Female rats with evidence of sperm in a vaginal smear or copulatory plug were designated as GD 0 and assigned to individual housing. Parental females were evaluated for length of gestation, fertility index, gestation index, number and sex of offspring per litter, number of implantation sites, general condition of the dam and litter during the postpartum period, litter size and viability, viability index, lactation index, percent survival, and sex ratio. Maternal behavior of the dams was recorded on LDs 1, 5, 8, 15, and 22.

F0 generation animals were sacrificed by carbon dioxide asphyxiation (day 106 to 110 of the study for male rats, i.e., after completion of the cohabitation period; and LD 22 for female rats), necropsied, and examined for gross lesions. Gross necropsy included examination of external surfaces and orifices, as well as internal examination of tissues and organs. Individual organs were weighed and organ-to-body weight and organ-to-brain weight ratios were calculated for the brain, kidneys, spleen, ovaries, testes, thymus, liver, adrenal glands, pituitary, uterus with oviducts and cervix, left epididymis (whole and cauda), right epididymis, prostate and seminal vesicles, (with coagulating glands and with and without fluid). Tissues retained in neutral buffered 10% formalin for possible histological evaluation included the pituitary, adrenal glands, vagina, uterus, with oviducts, cervix and ovaries, right testis, seminal vesicles, right epididymis, and prostate. Histological examination was performed on tissues from 10 randomly selected rats per sex from the control and high dosage groups. All gross lesions were examined histologically. All F0 generation rats that died or appeared moribund were also examined.

Histological examination of the reproductive organs in the low- and mid-dose groups was conducted in rats that exhibited reduced fertility by either failing to mate, conceive, sire, or deliver healthy offspring; or for which estrous cyclicity or sperm number, motility, or morphology were altered. Sperm number, motility, and morphology were evaluated in the left cauda epididymis of F0 generation male rats; testicular spermatid concentrations were evaluated in the left testis. The number and distribution of implantation sites were recorded in F0 generation female rats. Rats that did not deliver a litter were sacrificed on GD 25 and examined for pregnancy status. Uteri of apparently nonpregnant rats were examined to confirm the absence of implantation sites. A gross necropsy of the thoracic, abdominal and pelvic viscera was performed. Female rats without a confirmed mating date that did not deliver a litter were sacrificed on an estimated day 25 of gestation.

At scheduled sacrifice, after completion of the cohabitation period in F0 male rats and on LD 22 in F0 female rats, blood samples (10 males and 10 females each for the 10 and 30 mg/kg/day dose groups; 3 males and 3 females for the control group) were collected and frozen for future analysis. The methods section cites that liver samples were also collected, but no other details were provided and the results did not appear to be available at the time of the report.

The F1 generation pups in each litter were counted once daily. Physical signs (including variations from expected lactation behavior and gross external physical anomalies) were recorded for the pups each day. Pup body weights were recorded on LDs 1, 5, 8, 15 and 22. On LD 12, all F1 generation male pups were examined for the presence of nipples. Pups that died before examination of the litter for pup viability on LD 1 were evaluated for vital status at birth. Pups found dead on LDs 2 to 22 were examined for gross lesions and for the cause of death. All F1 generation rats were weaned on LD 22 based on observed growth and viability of these pups.

At weaning (LD 22), two F1 generation pups per sex per litter per group (60 male and 60 female pups per group) were selected for continued evaluation, resulting in 600 total rats (300 rats per sex) assigned to the five dosage groups. At least two male pups and two female pups per litter, when possible, were selected. F1 generation pups not selected for continued observation for sexual maturation were sacrificed. Three pups per sex per litter were examined for gross lesions. Necropsy included a single cross-section of the head at the level of the frontal-parietal suture and examination of the cross-sectioned brain for apparent hydrocephaly. The brain, spleen and thymus from one of the three selected pups per sex per litter were weighed and the brain, spleen, and thymus from the three selected pups per sex per litter were retained for possible histological evaluation. All remaining pups were discarded without further examination.

The F1 generation rats were given the same dosage level of the test substance and in the same manner as their respective F0 generation sires and dams. Dosages were given once daily, beginning at weaning and continuing until the day before sacrifice. F1 generation female rats were examined for age of vaginal patency, beginning on day 28 postpartum (LD 28). F1 generation male rats were evaluated for age of preputial separation, beginning on day 39 postpartum (LD 39). Body weights were recorded when rats reached sexual maturation.

Estrous cycling was evaluated daily by examination of vaginal cytology beginning 21 days before the scheduled cohabitation period and continuing until confirmation of mating by the presence of sperm in a vaginal smear or confirmation of a copulatory plug. On the day of scheduled sacrifice, the stage of the estrous cycle was assessed.

A table of random units was used to assign F1 generation rats to cohabitation, one male rat per female rat. If random assignment to cohabitation resulted in the pairing of F1 generation siblings, an alternate assignment was made. The cohabitation period consisted of a maximum of 14 days.

Body weights of the F1 generation male rats were recorded weekly during the postweaning period and on the day of sacrifice. Body weights of the F1 generation female rats were recorded weekly during the postweaning period to cohabitation, and on DGs 0, 7, 10, 14, 18, 21 and 25 (if necessary) and on LDs 1, 5, 8, 11, 15 and 22. Food consumption values for the F1 generation male rats were recorded weekly during the dosage period. Food consumption values for the F1 generation female rats were recorded weekly during the postweaning period to cohabitation, on GDs 0, 7, 10, 14, 18, 21 and 25 and on LDs 1, 5, 8, 11 and 15. Because pups begin to consume

maternal food on or about LD 15, food consumption values were not tabulated after LD 15.

At scheduled sacrifice, the F1 animals were subjected to gross necropsy, and selected organs were weighed and examined histologically as described above for the F0 animals. Sperm analyses were also conducted as described for the F0 animals.

F2 generation litters were examined after delivery to identify the number and sex of pups, stillbirths, live births and gross alterations. Each litter was evaluated for viability at least twice each day of the 22-day postpartum period. Dead pups observed at these times were removed from the nesting box. Anogenital distance was measured for all live F2 generation pups on LDs 1 and 22.

Parental Males (F0)

One F0 male rat in the 30 mg/kg/day dose group was sacrificed on day 45 of the study due to adverse clinical signs (emaciation, cold-to-touch, and decreased motor activity). Necroscopic examination in that animal revealed a pale and tan liver, and red testes. All other F0 generation male rats survived to scheduled sacrifice. Statistically significant increases in clinical signs were also observed in male rats in the high-dose group that included dehydration, urine-stained abdominal fur, and ungroomed coat.

Significant reductions in body weight and body weight gain were reported for most of the dosage period and continuing until termination of the study in the 3, 10, and 30 mg/kg/day dose groups. Absolute food consumption values were also significantly reduced during these periods at the 30 mg/kg/day dose group, while significant increases in relative food consumption values were observed in the 3, 10, and 30 mg/kg/day within those same periods.

No treatment-related effects were reported at any dose level for any of the mating and fertility parameters assessed, including numbers of days to inseminate, numbers of rats that mated, fertility index, numbers of rats with confirmed mating dates during the first and second week of cohabitation, and numbers of pregnant rats per rats in cohabitation. At necropsy, none of the sperm parameters evaluated (sperm number, motility, or morphology) were affected by treatment at any dose level.

At necropsy, statistically significant reductions in terminal body weights were seen at 3, 10, and 30 mg/kg/day. Absolute weights of the left and right epididymides, left cauda epididymis, seminal vesicles (with and without fluid), prostate, pituitary, left and right adrenals, spleen, and thymus were also significantly reduced at 30 mg/kg/day. The absolute weight of the seminal vesicles without fluid was significantly reduced in the 10 mg/kg/day dose group. The absolute weight of the liver was significantly increased in all dose-groups. Kidney weights were significantly increased in the 1, 3, and 10 mg/kg/day dose groups, but significantly decreased in the 30 mg/kg/day group. All organ weight-to-terminal body weight and ratios were significantly increased in all treated groups. Organ weight-to-brain weight ratios were significantly reduced for some organs at the high dose group, and significantly increased for other organs among all

treated groups.

No treatment-related effects were seen at necropsy or upon microscopic examination of the reproductive organs, with the exception of increased thickness and prominence of the zona glomerulosa and vacuolation of the cells of the adrenal cortex in the 10 and 30 mg/kg/day dose groups.

Serum analysis for the F0 generation males sampled at the end of cohabitation showed that PFOA was present in all samples tested, including controls. Control males had an average concentration of 0.0344± 0.0148 ug/ml PFOA. Treated males had 51.1±9.30 and 45.3±12.6 ug/ml, respectively for the 10 and 30 mg/kg/day dose groups.

Parental Females (F0)

No treatment-related deaths or adverse clinical signs were reported in parental females at any dose level. No treatment-related effects were reported for body weights, body weight gains, and absolute and relative food consumption values.

There were no treatment-related effects on estrous cyclicity, mating or fertility parameters. None of the natural delivery and litter observations were affected by treatment, that is, the numbers of dams delivering litters, the duration of gestation, the averages for implantation sites per delivered litter, the gestation index (number of dams with one or more liveborn pups/number of pregnant rats), the numbers of dams with stillborn pups, dams with all pups dying, liveborn and stillborn pups viability index, pup sex ratios, and mean birth weights were comparable to controls among all treated groups.

Necropsy and histopathological evaluation were also unremarkable. Terminal body weights, organ weights, and organ-to-terminal body weight ratios were comparable to control values for all treated groups, except for kidney and liver weights. The weights of the left and right kidney, and the ratios of these organ weights-to-terminal body weights, and of the left kidney weight-to-brain weight were significantly reduced at the highest dose of 30 mg/kg/day. The ratio of liver weights-to-terminal body weight was also significantly reduced at 3 and 10 mg/kg/day.

Results of the serum analysis in F0 generation females sampled on LD 22 showed that PFOA was present in all samples tested, except in controls where the level was below the limits of quantitation (0.00528ugm/l). Treated females had an average concentration of 0.37±0.0805 and 1.02±0.425 ug/ml, respectively for the 10 and 30 mg/kg/day dose groups.

F1 Generation - Males

No effects were reported at any dose level for the viability and lactation indices. No differences between treated and control groups were noted for the numbers of pups surviving per litter, the percentage of male pups, litter size and average pup body weight per litter at birth. Pup body weight on a per litter basis (sexes combined) was significantly reduced in the 30 mg/kg/day

group on days 1, 5, and 8 of lactation. Of the pups necropsied at weaning, no statistically significant, treatment-related differences were observed for the weights of the brain, spleen and thymus and the ratios of these organ weights to the terminal body weight and brain weight.

Significant increases in treatment-related deaths (7 animals total) were reported in F1 males in the high dose group of 30 mg/kg/day. One rat was moribund sacrificed on day 39 postweaning and another was found dead on day 107 postweaning, but the majority of the F1 male rats were found dead on days 2-4 postweaning.

Statistically significant increases in clinical signs of toxicity were also observed in F1 males during most of entire postweaning period. These signs included an increased incidence of annular constriction of the tail at all doses, with statistical significance at the 1, 10, and 30 mg/kg/day; a significant increase at 10 and 30 mg/kg/day in the number of male rats that were emaciated; and a significant increase in the incidence of urine-stained abdominal fur, decreased motor activity, and abdominal distention at 30 mg/kg/day.

Body weights and body weight gains were statistically significantly reduced prior to and during cohabitation and during the entire dosing period in all treated groups. Statistically significant reductions in body weights were observed at 10 and 30 mg/kg/day during days 8-15, 22-29, 29-36, 43-50, and 50-57 postweaning. Body weight gains were also significantly reduced in the 30 mg/kg/day group on days 1-8, 15-22, 36-43, 57-64, and 64-70 postweaning. Statistically significant, dose-related reductions in body weight gains were observed for the entire dosage period (days 1-113 postweaning). Absolute food consumption values were significantly reduced at 10 and 30 mg/kg/day during the entire precohabitation period (days 1-70 postweaning), while relative food consumption values were significantly increased.

Statistically significant ($p \leq 0.01$) delays in sexual maturation (the average day of preputial separation) were observed in high-dose animals versus concurrent controls (52.2 days of age versus 48.5 days of age, respectively).

No apparent effects were observed on any of the mating or fertility parameters including fertility and pregnancy indices (number of pregnancies per number of rats that mated and rats in cohabitation, respectively), the number of days to inseminate, the number of rats that mated, and the number of rats with confirmed mating dates during the first week. No statistically significant, treatment-related effects were observed on any of the sperm parameters (motility, concentration, or morphology).

Necroscopic examination revealed statistically significant treatment-related effects at 3, 10, and 30 mg/kg/day ranging from tan areas in the lateral and median lobes of the liver to moderate to slight dilation of the pelvis of one or both kidneys.

Statistically significant, dose-related decreases in terminal body weights of parental F1 males were observed. The absolute weights of the liver were significantly increased and the absolute weights of the spleen were significantly decreased at all treated groups. The absolute weights of

the left and/or right kidneys were significantly increased in the 1 and 3 mg/kg/day dose groups and significantly decreased in the 30 mg/kg/day dose group. The absolute weight of the thymus was also significantly decreased in the 10 and 30 mg/kg/day dose groups. The absolute weight of the prostate, brain and left adrenal gland were significantly decreased in the 30 mg/kg/day dosage group. The ratios of the weights of the seminal vesicles, with and without fluid, liver and left and right kidneys to the terminal body weights were significantly increased in all treated groups. The ratios of the weights of the left testis, with and without the tunica albuginea and the right testis to the terminal body weight, were significantly increased at 3 mg/kg/day and higher. The ratios of the weights of the left epididymis, left cauda epididymis, right epididymis and brain to the terminal body weight were significantly increased at 10 mg/kg/day and higher. The ratios of the weight of the seminal vesicles with fluid to the brain weight were increased at 1 mg/kg/day and higher, with statistical significance at 1 and 10 mg/kg/day. The ratios of the liver weight-to-brain weight were significantly increased in the 1 mg/kg/day and higher dosage groups, and the ratios of the left and right kidney weights-to-brain weight were significantly increased in all treated groups. The ratios of the spleen weight-to-brain weight were significantly decreased at 1 mg/kg/day and higher, and the ratios of the thymus weight-to-brain weight were significantly decreased at 10 and 30 mg/kg/day. The ratios of the left and right testes weight-to-brain weight were increased in the 3 mg/kg/day and higher dosage groups. These ratios were significantly increased at 10 mg/kg/day (right testis only) and 30 mg/kg/day.

Histopathologic examination of the reproductive organs was unremarkable; however, treatment-related microscopic changes were observed in the adrenal glands of high-dose animals (cytoplasmic hypertrophy and vacuolation of the cells of the adrenal cortex) and in the liver of animals treated with 3, 10, and 30 mg/kg/day (hepatocellular hypertrophy). No other treatment-related effects were reported.

F1 Generation – Females

No effects were reported at any dose level for the viability and lactation indices. No differences between treated and control groups were noted for the numbers of pups surviving per litter, the percentage male pups, litter size and average pup body weight per litter at birth. Pup body weight on a per litter basis (sexes combined) was significantly reduced in the 30 mg/kg/day group on days 1, 5, and 8 of lactation.

At 30 mg/kg/day, one pup from one dam died prior to weaning on lactation day 1 (LD1). Additionally, on lactation days 6 and 8, statistically significant increases in the numbers of pups found dead were observed at 3 and 30 mg/kg/day. According to the study authors, this was not considered to be treatment related because they did not occur in a dose-related manner and did not appear to affect any other measures of pup viability that included, numbers of surviving pups per litter and live litter size at weighing. An independent statistical analysis was conducted by US EPA (2002b). No significant differences were noted between dose groups and there was no significant trend.

Of the pups necropsied at weaning, no statistically significant, treatment-related differences were observed for the weights of the brain, spleen and thymus and the ratios of these organ weights to the terminal body weight and brain weight.

An increase in treatment-related mortality (6 animals total) was observed in F1 females on postweaning days 2-8 at the highest dose of 30 mg/kg/day. No adverse clinical signs of treatment-related toxicity were reported for any dose level during any time of the study period.

Statistically significant decreases in body weights and body weight gains were observed in high-dose animals on days 8, 15, 22, 29, 50, and 57 postweaning, during precohabitation (recorded on the day cohabitation began, when F1 generation rats were 92-106 days of age), and during gestation and lactation. Decreases in absolute food consumption were observed during days 1-8, 8-15 postweaning during precohabitation and during gestation and lactation in animals treated with 30 mg/kg/day. Relative food consumption values were comparable across all treated groups.

Statistically significant ($p \leq 0.01$) delays in sexual maturation (the average day of vaginal patency) were observed in high-dose animals versus concurrent controls (36.6 days of age versus 34.9 days of age, respectively).

Prior to mating, the study authors noted a statistically significant increase in the average numbers of estrous stages per 21 days in high-dose animals (5.4 versus 4.7 in controls). For this calculation, the number of independent occurrences of estrus in the 21 days of observation was determined. This type of calculation can be used as a screen for effects on the estrous cycle, but a more detailed analysis should then be conducted to determine whether there is truly an effect. 3M Company (2002) recently completed an analysis that showed there were no effects on the estrous cycle; there were no differences in the number of females with $\geq$ 3 days of estrus or with $\geq$ 4 days of diestrus in the control and high dose groups. Analyses conducted by the US EPA (2002a) also demonstrated that there were no differences in the estrous cycle among the control and high dose groups. The cycles were evaluated as having either regular 4-5 day cycles, uneven cycling (defined as brief periods with irregular pattern) or periods of prolonged diestrus (defined as 4-6 day diestrus periods) extended estrus (defined as 3 or 4 days of cornified smears), possibly pseudopregnant, (defined as 6-greater days of leukocytes) or persistent estrus (defined as 5-or greater days of cornified smears). The two groups were not different in any of the parameters measured. Thus, the increase in the number of estrous stages per 21 days that was noted by the study authors is due to the way in which the calculation was done, and is not biologically meaningful.

No effects on any of the mating and fertility parameters (numbers of days in cohabitation, numbers of rats that mated, fertility index, rats with confirmed mating dates during the first week of cohabitation and number of rats pregnant per rats in cohabitation).

All natural delivery observations were unaffected by treatment at any dose level. Numbers of dams delivering litters, the duration of gestation, averages for implantation sites per delivered

litter, the gestation index (number of dams with one or more liveborn pups/number of pregnant rats), the numbers of dams with stillborn pups, dams with all pups dying and liveborn and stillborn pups were comparable among treated and control groups.

No treatment-related effects were observed on terminal body weights. The absolute weight of the pituitary and the ratios of the pituitary weight-to-terminal body weight and to the brain weight were significantly decreased at 3 mg/kg/day and higher, but did not show a dose-response. No other differences were reported for the absolute weights or ratios for other organs evaluated. No treatment-related effects were reported following necroscopic and histopathologic examinations.

F2 Generation Offspring

No treatment-related adverse clinical signs were observed at any dose level. Likewise, no treatment-related effects were reported following necroscopic examination, with the exception of no milk in stomach in pups that were found dead. The numbers of pups found either dead or stillborn did not show a dose-response (3/28, 6/28, 10/28, 10/28, and 6/28 in 0, 1, 3, 10, and 30 mg/kg/day dose groups, respectively) and therefore were unlikely related to treatment.

No effects were reported at any dose level for the viability and lactation indices. No differences between treated and control groups were noted for the numbers of pups surviving per litter, the percentage of male pups, litter size and average pup body weight per litter when measured on LDs 1, 5, 8, 15,or 22. Anogenital distances measured for F2 male and female pups on LDs 1 and 22 were also comparable among the five dosage groups and did not differ significantly.

Statistically significant increases ($p \leq 0.01$) in the number of pups found dead were observed on lactation day 1 at the 3 and 10 mg/kg/day dosage groups. According to the study authors, this was not considered to be treatment related because they did not occur in a dose-related manner and did not appear to affect any other measures of pup viability that included, numbers of surviving pups per litter and live litter size at weighing. An independent statistical analysis was conducted by US EPA (2002b). No significant differences were observed between dose groups and there was no significant trend.

Terminal body weights in F2 pups were not significantly different from controls. Absolute weights of the brain, spleen and thymus and the ratios of these organ weights-to-terminal body weight and to brain weight were also comparable among treated and control groups.

Conclusions

Dosing with APFO at 30 mg/kg/day appeared to delay the onset of sexual maturation in both male and female F1 offspring. The authors of the study contend that the delays in sexual maturation (preputial separation or vaginal patency) observed in high-dose animals are due to the fact that these animals have a decreased gestational age, a variable which they have defined as the time in days from evidence of mating in the F0 generation until evidence of sexual

maturation in the F1 generation. The authors state that gestational age appeared to be decreased in high-dose animals at the time of acquisition (the time when sexual maturation was reached), which they believe meant the animals in that group were younger and more immature than the control group, in which there was no significant difference in sexual maturation.

In order to test this hypothesis, the authors covaried separately the decreases in body weight and in gestational age with the delays in sexual maturation in order to determine whether or not body weights and gestational age were a contributing factor. When the body weight was covaried with the time to sexual maturation, the time to sexual maturation showed a dose related delay that was statistically significant at the $p \leq 0.05$. This suggests that the delay in sexual maturation was partly related to body weight, but not entirely. When gestational age was covaried with the time to sexual maturation, there was no significant difference in the time of onset of sexual maturation between controls and high-dose animals. This indicates that the effect of delayed sexual maturation could possibly be attributed to decreased gestational age.

While it is known and commonly accepted that changes in the body weights of offspring can affect the time to sexual maturation, whether or not gestational age, as defined by the authors, also affects the time of acquisition is purely speculative, especially since there was no data provided by the authors to support this relationship. Additionally, covarying gestational age with time to sexual maturation is problematic from a statistical standpoint. Since there was no significant change in the length of gestation at 30 mg/kg/day, based on the authors' definition of 'gestational age', the decreases in gestational age would have to be due mostly to changes in time to sexual maturation. Therefore, sexual maturation is essentially being covaried with itself. Still, even if a relationship between gestational age and time to sexual maturation were shown, it merely offers an explanation for the observed delays in sexual maturation in high-dose animals, but does not diminish its significance.

Therefore, under the conditions of the study, the LOAEL for F0 parental males is considered to be 1 mg/kg/day, the lowest dose tested, based on significant increases in the liver and kidney weights-to-terminal body weight and to brain weight ratios. A NOAEL for the F0 parental males could not be determined since treatment-related effects were seen at all doses tested.

The NOAEL and LOAEL for F0 parental females are considered to be 10 and 30 mg/kg/day, respectively, based on significant reductions in kidney weight and kidney weight-to-terminal body weight and to brain weight ratios observed at the highest dose.

The LOAEL for F1 generation males is considered to be 1 mg/kg/day, based on significant decreases in body weights and body weight gains, and in terminal body weights; and significant changes in absolute liver and spleen weights and in the ratios of liver, kidney, and spleen weights-to-brain weights; and based on significant, dose-related reductions in body weights and body weight gains observed prior to and during cohabitation and during the entire dosing period. A NOAEL for the F1 males could not be determined since treatment-related effects were seen at all doses tested.

The NOAEL and LOAEL for F1 generation females are considered to be 10 and 30 mg/kg/day, respectively, based on statistically significant increases in postweaning mortality, delays in sexual maturation (time to vaginal patency), decreases in body weight and body weight gains, and decreases in absolute food consumption, all observed at the highest dose tested.

The NOAEL for the F2 generation offspring was considered to be 30 mg/kg/day. No treatment-related effects were observed at any doses tested in the study. However, it should be noted that the F2 pups were sacrificed at weaning, and thus it was not possible to ascertain the potential post-weaning effects that were noted in the F1 generation.

3.8 Carcinogenicity Studies in Animals

3.8.1 Cancer Bioassays

The carcinogenic potential of APFO has been investigated in a two-year feeding study in rats (3M, 1987). In this study, groups of 50 male and 50 female Sprague-Dawley (Crl:CD BR) rats were fed diets containing 0, 30 or 300 ppm APFO for two years. Groups of 15 additional rats per sex were fed 0, or 300 ppm APFO and evaluated at the one-year interim sacrifice. The mean actual test article consumption was: males, 1.3 and 14.2 mg/kg/day; females, 1.6 and 16.1 mg/kg/day for the low and high-dose groups, respectively.

There was a dose-related decrease in body weight gain in the male rats and to a lesser extent, in the female rats as compared to the controls; the decreases were statistically significant in the high-dose groups of both sexes. The body weight changes are treatment related since feed consumption was actually increased (rather than decreased). There were no differences in mortality between the treated and untreated groups; the survival rates at the end of 104 weeks for the control, low-, and high-dose groups were: male, 70%, 72% and 88%; females, 50%, 48% and 58%. The only clinical sign observed was a dose-related increase in ataxia in the female rats; the incidences in the control, low- and high-dose groups were: 4%, 18% and 30%. Significant decreases in red blood cell counts, hemoglobin concentrations and hematocrit values were observed in the high-dose male and female rats as compared to control values. Clinical chemistry changes indicative of liver toxicity included increases in alanine aminotransferase (ALT), aspartate aminotransferase (AST) and alkaline phosphatase (AP) in both treated male groups from 3-18 months, but only in the high-dose males at 24 months. Increases in relative liver and kidney weights were noted in both high-dose male and female rats. Significant non-neoplastic lesions were seen primarily in the liver and testis; there were increases in the incidence of liver masses, hyperplastic nodules and foci, and in testicular masses in the high-dose male group. Other liver toxic effects include dose-related increases in the incidence of diffuse hepatomegalocytosis, cystoid degeneration, and portal mononuclear cell infiltration in both male and female treated groups; these increases were statistically significant in the high-dose males. A statistically significant, dose-related increase in the incidence of ovarian tubular hyperplasia was found in female rats; the incidence of this lesion in the control, low-, and high-dose groups was 0%, 14%, and 32%, respectively. Based on these toxic effects, the high dose selected in this study appears to have reached the Maximum Tolerated Dose (MTD). Based on

decreased body weight gain, increased liver and kidney weights and toxicity in the hematological and hepatic systems, the LOAEL for male and female rats is 300 ppm. [Based on increases in the incidence of ataxia and ovarian tubular hyperplasia, the LOAEL for female rats is 30 ppm.]

At the termination of the study, a slight increase in the incidence of various neoplasms (tumors of the liver, testis, thyroid, adrenal and mammary glands, etc.) was seen in the treated animals. Among them, the increased incidences of testicular (Leydig) cell adenomas in the high-dose male rats, and of mammary fibroadenoma in both groups of female rats were statistically significant ($P< 0.05$) as compared to the concurrent controls. The incidence of the Leydig cell tumors (LCT) in the control, low- and high-dose males was 0/50 (0%), 2/50 (4%) and 7/50 (14%), respectively; the respective incidences of mammary fibroadenoma in the female groups were 11/50 (22%), 21/50 (42%) and 24/50 (48%). The increases are also statistically significant as compared to the historical control incidences (LCT, 0.82%; mammary fibroadenoma, 19.0%) observed in 1,340 male and 1,329 female Sprague-Dawley control rats used in 17 carcinogenicity studies (Chandra et al., 1992). The spontaneous incidence of LCT in 2-year old Sprague-Dawley rats in other studies was reported to be approximately 5% (*cited in*: Clegg et al., 1997). Therefore, under the conditions of this study, APFO is carcinogenic in Sprague-Dawley rats, inducing Leydig cell tumors in the male rats and mammary fibroadenomas in the female rats.

The induction of Leydig cell tumors was confirmed in a follow-up 2-year mechanism study of PFOA in male Sprague-Dawley (CD) rats at a dietary level of 300 ppm (Cook et al., 1994; Biegel et al. 2001). A significantly increased LCT incidence was observed in the treated rats (8/76, 11%) as compared to the controls (0/80, 0%). In addition, PFOA also caused significantly increased incidences of liver tumors and pancreatic acinar cell tumors. The incidences of liver adenomas in the control and treated groups were 2/80 (3%) and 10/76 (13%), respectively, whereas those for the pancreatic acinar cell adenomas were 0/80 (0%) and 7/76 (9%). There was one pancreatic acinar cell caricinoma in 76 of the treated rats and none in 80 controls. The incidence of combined pancreatic acinar cell adenoma/carcinoma in the treated rats (8/76, 11%) was also significantly increased as compared to the controls (0/80, 0%).

PFOA has also been shown to promote liver carcinogenesis in rodents (Abdellatif et al.,1991; Nilsson et al., 1991).

3.8.2 Mode of Action Studies

The mechanism(s) of toxicological/carcinogenic action of PFOA is not clearly understood. PFOA was not mutagenic in the Ames test using five strains of *Salmonella typhimurium*, or in an assay with *Saccharomyces cerevisiae* (Griffith and Long, 1980). Short-term genotoxicity assays appear to suggest that PFOA is not a DNA-reactive compound. However, when tested with metabolic activation, PFOA induced significant increases in chromosomal aberrations and in polyploidy in CHO cells (Murli, 1996). The significance of these genotoxic effects is unclear. Available data appear to indicate that the induction of tumors by PFOA is due to a non-genotoxic mechanism, involving activation of receptors and perturbations of the endocrine system. The

Agency is currently examining these postulated modes of action in detail. The following summaries are not meant to be a detailed review of the literature, but simply summarize the current scientific evidence.

3.8.2.1 Liver Tumors

It has been well documented that APFO is a potent peroxisome proliferator, inducing peroxisome proliferation in the liver of rats and mice (*e.g.*, Ikeda et al., 1985; Pastoor et al., 1987; Sohlenius et al., 1992). A sex-related difference in the induction of liver peroxisome proliferation exists in rats (Kawashima et al., 1989), but not in mice (Sohlenius et al., 1992). The higher induction of liver peroxisome proliferation in male rats was shown to be strongly dependent on the sex hormone testosterone (Kawashima et al., 1989). Like many other peroxisome proliferators, APFO has also been shown to cause hepatomegaly (an early biomarker of peroxisome proliferator hepatocarcinogenesis) in rats (Takagi, et al., 1992; Cook, 1994) and mice (Kennedy, 1987), and induce oxidative DNA damage in liver of rats (Takagi et al., 1991). The totality of these data appears to suggest that the liver toxicity and carcinogenicity of APFO may be related to induction of peroxisome proliferation. Meanwhile, estrogen has been shown to promote hepatocarcinogenesis in rats (Yager and Yager, 1980; Cameron et al., 1982); an increase in estrogen levels after APFO exposure (discussed below) may also play a role in hepatocarcinogenesis in rats. Recently, IARC (1995) concluded that the liver tumors induced in rodents by PPAR-alpha agonists are unlikely to be operative in humans based on our current understanding of the animal mode of action.

3.8.2.2 Leydig Cell Tumors

A large number of non-genotoxic compounds of diverse chemical structures have been reported to induce Leydig cell tumors (LCT) in rats, mice, or dogs. A review of the available information on LCT induction in animals led a workshop panel to classify these compounds into seven groups based on their modes of action (Clegg et al., 1997). The common theme in the mode of action for most compounds is that these compounds affect the hormonal control of Leydig cell growth by disrupting the hypothalamic-pituitary-testicular axis at various points that result in increasing the serum levels of luteinizing hormone (LH). It has been postulated that in addition to stimulating the production of testosterone, LH may also play a mitogenic role in the Leydig cells; a sustained increase in circulating LH levels and chronic stimulation of Leydig cells by growth-stimulating mediators such as IGF-1, TGF-β, leukotrienes and various free radicals can lead to LCT development (*rev. in*: Clegg et al., 1997).

A series of studies have been conducted to investigate the mechanism of tumor formation in male Sprague-Dawley (CD) rats exposed to APFO (Cook et al., 1992; Biegel et al., 1995; Liu et al., 1996). No significant increases in LH were seen in the rats after treatment of APFO at various dose levels for 14 days. However, serum and testicular levels of estradiol were significantly increased and testosterone levels were significantly decreased. It was postulated that the elevated estradiol levels may cause Leydig cell hyperplasia and tumor formation by acting as a mitogen and/or enhancing growth factor secretion; the transforming growth factor α

(TGF α), which binds to the epidermal growth factor (EGF) receptor and stimulated cell proliferation, for instance, has been detected in Leydig cells (Teerds et al., 1990). Subsequent experiments have shown that APFO increased the levels of estradiol by inducing cytochrome P450 XIX (aromatase), which converts testosterone to estradiol. Peroxisome proliferators are known to induce β-oxidation and cytochrome P-450 monooxygenases by binding to the peroxisome proliferation activation receptor α (PPAR α; a subfamily of steroid hormone receptors). It is believed that APFO induces cytochrome P450 XIX (aromatase) by binding to and activating the PPARα.

Although significant increases in LH were not seen in Sprague-Dawley rats after treatment of APFO in the 14 day-studies, it appears that increase in LH levels cannot be ruled out to be involved (in addition to increased estradiol level) in the induction of LCT by APFO. In these studies, significant increase in hepatic aromatase (which converts testosterone to estradiol) activities associated with decreased serum testosterone levels and increased estradiol levels were observed in the treated rats. Testosterone is synthesized and secreted by the Leydig cells, and is regulated by LH; testosterone and LH form a closed-loop feedback system in the HPT axis. In order to maintain adequate testosterone plasma levels, reduced testosterone levels (caused by increased aromatase activity) are expected to lead to increased LH levels through the negative feedback mechanism. It has been pointed out that increases in LH may not always be seen in all studies of chemicals for which the proposed mode of action calls for elevated LH, and that compensation may have occurred to restore homeostasis and inappropriate timing of sampling are some of the explanations for failing to detect changes in LH levels (Clegg et al., 1997).

3.8.2.3 Mammary Gland Tumors

Estradiol has also been shown to stimulate the secretion of TGF α by mammary epithelial cells and the overexpression of TGF α has been suggested as one possible factor in producing sustained cell proliferation of mammary tumor cells and the subsequent development of neoplasia (Liu et al., 1987). Hence, it is possible that the APFO-induced elevation of estradiol levels may also be responsible for the development of mammary fibroadenomas in Sprague Dawley rats in addition to LCT (discussed above). In fact, this is consistent with the mechanism by which spontaneous mammary neoplasms were developed in aging female Sprague Dawley rats. It has been demonstrated that the early appearance and high spontaneous incidence of mammary gland tumors in untreated, aging female Sprague-Dawley rats is due to increased exposure to endogenous estrogen and prolactin as a result of an accelerating effect on normal, age-related perturbations of the estrous cycle in this strain of rat (Cutts and Noble, 1964; Chapin et al., 1996).

3.8.2.4 Pancreatic Tumors

The mechanism by which APFO induced pancreatic acinar cell tumors is unknown. A number of other peroxisome proliferators also produce pancreatic acinar cell tumors in rats. Available data suggest that the pancreatic acinar cell tumors are related to an increase in serum cholecystokinin (CCK) level secondary to hepatic cholestasis (Cook et al., 1994; Obourn et al., 1997). CCK is a

growth factor that has been shown to stimulate normal, adaptive, and neoplastic growth of pancreatic acinar cells in rats (Longnecker, 1987). However, data on the role of CCK in pancreatic tumor formation are conflicting.

3.9 Immunotoxicology Studies in Animals

Four immunotoxicity studies of PFOA have been conducted in mice. The first of these studies was a feeding study in mice (Yang et al. 2000). For investigation of the effects of perfluorooctanoic acid (PFOA), and other peroxisome proliferators, on lymphoid organs, 0.02 % PFOA was administered to male C57Bl/6 mice in the diet for 2, 5, 7, or 10 days. At the end of the feeding period, mice were sacrificed and the liver, spleen, and thymus were dissected out and weighed. The effect of PFOA administration on the cellularity, cell surface phenotype, and cell cycle of thymocytes and splenocytes was determined. In addition, effects of exposure of thymocytes and splenocytes to PFOA in vitro were examined.

The results showed that administration of 0.02% PFOA for 2, 5, 7, or 10 days resulted in a significant increase, relative to controls, in liver weight, even at the earliest time point. Also, a decrease in body weight caused by PFOA administration was observed. Subsequently, by the day 5 administration period, significant decreases in thymus and spleen weight were detected. After administration of 0.02% PFOA for 7 days, significant decreases (85% and 80%, respectively) in the total number of thymocytes and splenocytes were observed. The results also showed that the number of thymocytes expressing both CD4 and CD8 decreased by 95%; the number expressing both CD4 and CD8 decreased by 57%; and the number expressing either CD4 or CD8 decreased by 64% and 72%, respectively. For the splenocytes, both T cells (CD3) and B cells (CD19) decreased by 75% and 86%, respectively. Also, significant decreases in both CD4 helper and CD8 cytotoxic splenic T cells were observed. Upon administration of 0.02% PFOA to mice for 7 days, thymocyte proliferation was also inhibited, as detected by cell cycle flow cytometry analyses. In vitro studies showed that there was spontaneous apoptosis occurring in splenocytes and thymocytes after 8 or 24 hours of culturing in the presence of varying concentrations (50, 100, or 200 M) of PFOA. However, PFOA did not significantly alter the cell cycle under these conditions.

In order to study mechanism (Yang et al. 2001) , another mouse feeding study was performed. In order to examine the dose dependency of the effects, C57Bl/6 mice received diets consisting of 0.001%-0.05% PFOA (w/w) for 10 days. For examining the time-course, a diet containing 0.02% PFOA was given for 2, 5, 7 or 10 days. Effects of withdrawal of PFOA were also studied.

The results showed that, at higher doses, a significant decrease, relative to controls, in body weight was observed, although no other apparent signs of toxicity such as sores, lethargy, and poor grooming were noticed. However, a significant decrease in total water intake was observed. Mice receiving dietary PFOA for 10 days experienced significant increases in liver weight and peroxisome proliferation, as measured by induction of acyl-CoA oxidase with lauroyl-CoA or palmitoyl-CoA as substrate. These increases started at the lowest dose and

reached their maximal values at a dose of 0.003-0.01%. In contrast, the weight decreases of the spleen and thymus began at a higher dose (0.01%) with no maximum reached with the doses given. The time course studies showed that increased liver weights and peroxisome proliferation were evident at the earliest time point examined. In contrast, significant thymus and spleen weight decreases required PFOA administration for a period of at least 5 days, following which the spleen weight remained constant while the thymus weight continued to decrease. However, upon prolonged treatment for one month, no further decreases in thymus and spleen weights were observed. In another set of experiments, animals received 0.02% PFOA for 7 days, and then they received normal chow for a period of 10 days. These recovery experiments showed that the animals rapidly recovered the body weight the second day after withdrawal of PFOA. However, the liver weight did not return to normal even after 10 days of recovery. Thymus recovery started on day 2 and was completed by day 10. The spleen weights returned to normal by day 2 post-withdrawal. In addition, the changes in thymus and spleen weight upon PFOA treatment and withdrawal paralleled the changes in total thymocyte and splenocyte counts. Furthermore, flow cytometry cell cycle experiments showed that the decrease in thymocyte number caused by PFOA treatment is due mainly to inhibition of thymocyte proliferation. In contrast, PFOA treatment caused no changes in the cell cycle of splenocytes.

A third feeding study (Yang et al. 2002a) was designed to examine the possible involvement of the peroxisome proliferator-activated receptor alpha (PPARα in the immunomodulation exerted by PFOA. This study made use of transgenic PPARα null mice, which are homozygous with regards to a functional mutation in the PPARα gene. These mice do not exhibit peroxisome proliferation or hepatomegaly and hepatocarcinogenesis even after exposure to peroxisome proliferators. These mice were fed a diet consisting of 0.02% PFOA (w/w) for 7 days. At the end of the feeding period, mice were sacrificed and the liver, spleen, and thymus were removed and weighed. The effect of PFOA on peroxisome proliferation, cell cycle, and lymphoproliferation was ascertained.

The results showed that, in contrast to wild-type mice, feeding PPARα null mice PFOA resulted in no significant decrease in body weight. However, increases in liver weight were still seen in PPARα null mice, suggesting that this is not a PPARα dependent process. As expected, peroxisome proliferation, as measured by fatty acid - oxidation, was totally lacking in PPARα null mice. Also in contrast to wild type mice, feeding PPAR(null mice PFOA resulted in no significant decrease in the weight of the spleen or the number of splenocytes. At the same time, the decrease in weight and cellularity of the thymus was attenuated, but not totally eliminated in the PPAR(null mice. In addition, the decreases in the size of the CD4+CD8+ population of thymus cells and the number of thymus cells in the S and G2/M phases of the cell cycle, which reflects inhibition of proliferation, observed in wild type mice administered PFOA were much less extensive in PPAR(null mice. Finally, in contrast to wild type mice, PFOA treatment caused no significant change in splenocyte proliferation in response to mitogens in PPAR(null mice.

A fourth feeding study (Yang et al. 2002b) was designed to examine the effects of PFOA on specific humoral immune responses in mice. For this study, 0.02 % PFOA was administered to

male C57Bl/6 mice for 10 days. Then the animals were examined, via plaque forming cell (PFC) and serum antibody assays, for their ability to generate an immune response to horse red blood cells (HRBCs). Ex vivo and in vitro splenic lymphocyte proliferation assays were also performed.

The results showed that mice fed normal chow responded to challenge with HRBCs with a strong humoral response, as measured by the PFC assay. In contrast, mice fed with PFOA responded to HRBC immunization with no increase in HRBC-specific PFCs, relative to unimmunized controls. However, in experiments where PFOA-treated mice received normal chow following HRBC immunization, there was a significant recovery of the numbers of specific PFCs stimulated. The suppression of the humoral immune response by PFOA was confirmed by analysis of the serum anti-HRBC response. In ex vivo experiments, splenocytes isolated from control mice responded to both ConA and LPS with lymphocyte proliferation, as measured by thymidine incorporation. However, treating mice with PFOA (0.02% for 7 days) attenuated the proliferation. In a set of in vitro experiments, PFOA (1- 200 M) added to the culture medium of splenocytes cultured from untreated mice did not cause an alteration of lymphocyte proliferation in response to LPS or ConA.

4.0 Hazards to the Environment

4.1 Introduction

The aquatic toxicity and hazard of APFO to aquatic organisms was assessed. This task was made more difficult by several problems discussed below. These problems complicated the task of determining if the ecotoxicity tests were valid and could be used in the assessment. Furthermore, these problems limited the confidence that could be placed on the toxicity test values, and thus in turn lowered the confidence of conclusions that could be drawn in assessing the inherent toxicity and hazard of APFO to aquatic organisms.

1) A variety of different APFOs with varying designations and lot numbers were tested. Generally, the ammonium salt or the tetrabutylammonium salt was tested. The exact composition and identification of impurities, which may affect toxicity, in each lot number used is not known.

2) A variety of testing laboratories conducted the APFO toxicity studies over a period of time from approximately 1974-1996. This situation served to increase overall test variability and thus made inter-laboratory comparisons more difficult.

3) Purity of the tested material, or percent test material and percent other material(s), was a major concern. Purity was not sufficiently characterized in these tests. In some tests it appeared that 100% test chemical was used; in others a chemical of lesser purity (approximately 85%) was used. Purity of test material does affect toxicity and should be taken into account when possible, by expressing toxicity on the same purity basis.

4) Water, an isopropanol solvent, or a combination of both were used with the test material in

many of the toxicity tests, for no obvious indicated reason. Solvents are mixed with the test material to make it miscible with the test dilution water before the test is begun. Solvents are used in tests where the concentrations of the test material are extremely low and a very small amount of test material must be added to the test chambers. It was not clear from the summaries of these studies why a solvent was used or was even found to be necessary. In fact, 3M summarized each test and stated "Data may not accurately relate toxicity of the test sample with that of the test substance." Thus, in those tests where 100% test material was not used, the toxicity values had to be adjusted to take into account the percent solvent(s), and to express the values on a 100% test chemical basis, so that the tests could be compared.

5) In all these toxicity tests only nominal test chemical concentrations were used. Measured test chemical concentrations are instead always recommended so that one can accurately determine the actual test chemical concentration to which the test organisms are exposed. If it is determined that the nominal concentrations are only, for example 50% of the measured concentrations, the toxicity values will have to accordingly be adjusted by 50%. Analytical measurements of chemical concentration should have been taken or made available. Then, recovery rates could have been determined, and physicochemical processes (e.g., hydrolysis, volatility) that might lower the actual concentrations to which the test organisms were exposed could have been taken into account. Nominals may be used when measured concentrations are taken and the relationship of both is known.

In order to proceed with any sort of environmental hazard review it was necessary to ignore these test limitations and to assume that the nominal concentrations were an "adequate" expression of the measured test chemical concentrations. Criteria for assessing degree of acute toxicity are based on well-established values (low is >100 mg/L; medium or moderate is >1<100 mg/L; high is <1 mg/L).

4.2 Acute Toxicity to Freshwater Species

Several species were tested to assess the acute toxicity of APFO; these included the fathead minnow (Pimephales promelas), bluegill sunfish (Lepomis machrochirus), water flea (Daphnia magna), and a green alga (Selenastrum capricornutum). The toxicity test endpoints have been adjusted to 100% test chemical and test results are presented in Tables 9 (organized by test substance) and 10 (organized by test species). Each value is related to a testing facility and reference.

Twelve tests were conducted with fathead minnows; 96-h LC50 values (based on mortality) ranged from 70 to 843 mg/L. It is unclear why this range is so wide. Assuming these studies are valid, and due to the limitations discussed above, these toxicity values indicate low toxicity. The two acute values for bluegill sunfish also indicate low toxicity (96-h LC50s of >420, and 569 mg/L).

Nine acute tests were conducted with daphnids and 48-h EC50 values (based on immobilization) ranged from 39 to >1000 mg/L. The lower values are indicative of moderate toxicity, but the

wide range makes interpretation difficult.

Seven tests were conducted with green algae; 96-h EC50 values (based on growth rate, cell density, cell counts, and dry weights) ranged from 1.2 to >666 mg/L (the Er50 cell density value of 1,000 mg/L is excluded from this discussion). The lower value indicates high to moderate toxicity, based on the acute criteria. The lower value would also be indicative of moderate toxicity, based on the chronic moderate criterion (.0.1<10 mg/L). A 14-d EC50 value of 43 mg/L, based on cell counts, for green algae was also calculated in one study. This is indicative of low chronic toxicity, based on the chronic criterion (10 mg/L). Green algae appeared to be the most sensitive test species in the 44% APFO test sample, daphnids were the next most sensitive, and fathead minnows were the least sensitive.

Table 9. Summary of Acute Ecological Toxicity Data for APFO (grouped by test substance)			
Test Organism	**Duration**	**Value (mg/L)***	**Reference**
Test Sample: APFO ammonium salt			
Fathead minnow (*Pimephales promelas*)	96-h LC50	70	3M Company, 1974a
	96-h LC50	766	3M Company, 1980a
	96-h LC50	301	3M Company, 1987c
	96-h LC50	740	Ward et al.,, 1995
Bluegill sunfish (*Lepomis machrochirus*)	96-h LC50	> 420	3M Company, 1978
	96-h LC50	569	3M Company, 1978
Water flea (*Daphnia magna*)	48-h EC50	126	3M Environmental Laboratory, 1982
	48-h EC50	> 1000	3M Environmental Laboratory, 1982
	48-h EC50	221	3M Company, 1987b
	48-h EC50	720	Ward et al., 1995
Green algae (*Selenastrum capricornutum*)	96-h EC50	310	Ward et al., 1995
	96-h EC50	1000	Ward et al., 1995
Bacteria (*Photobacterium phosphoreum*)	30-min EC50	870	3M Company, 1987a
	30-min EC50	730	3M Environmental Laboratory, 1996a
Activated sludge	7-min NOEC	1000	3M Company, 1980b
	30-min EC50	> 1000	3M Company, 1987d
Test Sample: APFO			
Fathead minnow (*Pimephales promelas*)	96-h LC50	440	3M Company, 1974b
	96-h LC50	843	3M Company, 1985
Test Sample: APFO ammonium salt in 50% water			
Fathead minnow (*Pimephales promelas*)	96-h LC50	>500	EnviroSystems, Inc., 1990a
	96-h NOEC	500	EnviroSystems, Inc., 1990a
Water flea (*Daphnia magna*)	48-h EC50	292	EnviroSystems, Inc., 1990b
Bacteria (*Photobacterium phosphoreum*)	30-min EC50	> 500	3M Environmental Laboratory, 1990a
Test Sample: APFO ammonium salt in 50% water, continued			
Activated sludge	30-min EC50	> 500	3M Environmental Laboratory, 1990b

Test Sample: APFO ammonium salt in 80% water			
Fathead minnow (*Pimephales promelas*)	96-h LC50	494	Ward et al., 1996a
Water flea (*Daphnia magna*)	48-h EC50	240	Ward et al., 1996c
Green algae (*Selenastrum capricornutum*)	96-h EC50	396	Ward et al., 1996b
	96-h EC50	> 666	Ward et al., 1996b
Bacteria (*Photobacterium phosphoreum*)	30 min EC50	630	3M Environmental Laboratory, 1996b
	30 min EC50	390	3M Environmental Laboratory, 1996c
Activated sludge	30-min EC50	>664	3M Environmental Laboratory, 1996d
Test Sample: APFO in 50% isopropanol			
Fathead minnow (*Pimephales promelas*)	96h LC50	140	T.R. Wilbury Laboratories, Inc., 1996a
Water flea (*Daphnia magna*)	48-h EC50	360	T.R. Wilbury Laboratories, Inc., 1996b
Green algae (*Selenastrum capricornutum*)	96-h EC50	90	T.R. Wilbury Laboratories, Inc., 1995
Test Sample: APFO (44%) in 27.9% water and 27.2% isopropanol			
Fathead minnow (*Pimephales promelas*)	96-h EC50	391	T.R. Wilbury Laboratories, Inc., 1995
Fathead minnow (*Pimephales promelas*)	96-h EC50	422	T.R. Wilbury Laboratories, Inc., 1995
Test Sample: APFO (44%) in 27.9% water and 27.2% isopropanol			
Water flea (*Daphnia magna*)	48-h EC50	41	Ward et al., 1995
Water flea (*Daphnia magna*)	48-h EC50	39	Ward et al., 1995
Green algae (*Selenastrum capricornutum*)	96-h EC50	2.1	Ward et al., 1995
Green algae (*Selenastrum capricornutum*)	96-h EC50	3.6	Ward et al., 1995
Green algae (*Selenastrum capricornutum*)	96-h EC50	1.2	Ward et al., 1995

*Values were adjusted to represent 100% active ingredient.

[A]These values may be inconsistent due to different diets tested.

Table 10. Summary of Ecological Toxicity Data for APFO (grouped by species)			
Test Organism	**Duration**	**Value (mg/L)**	**Reference**
Fathead minnow (*Pimephales promelas*)	96-h LC50	70[B]	3M Company, 1974a
	96-h LC50	766[B]	3M Company, 1980a
	96-h LC50	301[B]	3M Company, 1987c
	96-h LC50	440[C]	3M Company, 1974b
	96-h LC50	843[C]	3M Company, 1985
	96-h LC50	> 500[D]	EnviroSystems, Inc., 1990a
	96-h NOEC	500[D]	EnviroSystems, Inc., 1990a
	96-h LC50	494	Ward et al., 1996a
	96h LC50	140[F]	T.R. Wilbury Laboratories, Inc., 1996a
	30-day NOAEL	> 100[B]	EG&G Bionomics Aquatic Toxicology Laboratory, 1978
	96-h EC50	391[G]	T.R. Wilbury Laboratories, Inc., 1995
	96-h EC50	422[G]	T.R. Wilbury Laboratories, Inc., 1995
Bluegill sunfish (*Lepomis machrochirus*)	96-h LC50	> 420[B]	3M Company, 1978
	96-h LC50	569[B]	3M Company, 1978
Water flea (*Daphnia magna*)	48-h EC50	126[AB]	3M Environmental Laboratory, 1982
	48-h EC50	> 1000[AB]	3M Environmental Laboratory, 1982
	48-h EC50	221[B]	3M Company, 1987b
	48-h EC50	292[D]	EnviroSystems, Inc., 1990b
	48-h EC50	240	Ward et al., 1996c
	48-h EC50	360[F]	T.R. Wilbury Laboratories, Inc., 1996b
	21-day IC50	43[B]	3M Company, 1984
	21-day NOEC	22[B]	3M Company, 1984
	21-day NOEC	22[B]	3M Company, 1984
	48-h EC50	41[G]	Ward et al., 1995
	48-h EC50	39[G]	Ward et al., 1995

Table 10. Summary of Ecological Toxicity Data for APFO (grouped by species)			
Test Organism	**Duration**	**Value (mg/L)**	**Reference**
Green algae (*Selenastrum capricornutum*)	96-h EC50	396	Ward et al., 1996b
	96-h EC50	>666[E]	Ward et al., 1996b
	96-h EC50	90[F]	T.R. Wilbury Laboratories, Inc., 1995
	14-day EC50	43[B]	Elnabarawy, 1981
	96-h EC50	2.1[G]	Ward et al., 1995
	96-h EC50	3.6[G]	Ward et al., 1995
	96-h EC50	1.2[G]	Ward et al., 1995
Bacteria (*Photobacterium phosphoreum*)	30-min EC50	870[B]	3M Company, 1987a
	30-min EC50	730[B]	3M Environmental Laboratory, 1996a
	30-min EC50	>500[D]	3M Environmental Laboratory, 1990a
	30 min EC50	630	3M Environmental Laboratory, 1996b
	30 min EC50	390	3M Environmental Laboratory, 1996c
Activated sludge	7-min NOEC	1000[B]	3M Company, 1980b
	30-min EC50	>1000[B]	3M Company, 1987d
	30-min EC50	> 500[D]	3M Environmental Laboratory, 1990b
	30-min EC50	> 664[E]	3M Environmental Laboratory, 1996d

*Values were adjusted to represent 100% active ingredient.
[A]These values may be inconsistent due to different diets tested.
[B]Tested substance was APFO ammonium salt.
[C]Tested substance was APFO
[D]Tested substance was APFO ammonium salt in 50% water.
[E]Tested substance was APFO ammonium salt in 80% water.
[F]Tested substance was APFO in 50% isopropanol.
[G]Test Sample: APFO (44%) in 27.9% water and 27.2% isopropanol

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