GROUND WATER SIMULATION STUDY
PHASE IV
BAY PARK, TOPSHAM, MAINE

A report to:
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1.0 Introduction and Purpose

As described in my Dec. 1978 study, a low yield sandy aquifer extends from the northern boundary of Bay Park southwest to Foreside Road and the Androscoggin River. Ground water flow is generally southwestward from Bay Park toward Foreside Road. The State Subdivision Law and the Site Location of Development Act and regulations require that developments demonstrate that they will not adversely affect the natural environmental, including ground water quality and quantity.

The purpose of this report is to evaluate the ground water impact of the proposed Phase IV subdivision of Bay Park in Topsham, Maine, that is proposed by Lewis Stuart. Stuart has previously developed Phase I and part of Phase II (Fig. 1). I prepared a report dated Dec. 1978, "The Relationship of Bay Park Development to Regional Ground Water Aquifers," which discussed the impact of Phase II on the local surficial and bedrock aquifers. That report reached its conclusions by the use of a simple averaging ground water model. The Maine Department of Environmental Protection recently granted conditional approval of Phase III, for which I had prepared the Oct. 1980 report entitled "Ground Water Simulation Study, Bay Park, Topsham." An Addendum #1 was added to this report in Dec. 1980. The 1980 studies were conducted with the aid of a computerized ground water simulation model, which was used to document the ground water impact of Phase I, II, and III in conjunction with existing development in the area. This report summarizes additional ground water simulation studies and concludes that the Phase IV development, in conjunction with fully-developed Phases I, II, III and other existing development will not cause an unacceptable impact on the ground water quality or quantity outside of the land controlled by Lewis Stuart.

2.0 Methods

The ground water impact study must do the following: a) identify pollutant sources and background water quality, b) identify the applicable geology, c) identify the ground water hydrology, d) simulate ground water flow systems and pollution dispersion, and e) identify the applicable criteria for determining whether or not the development's impact can be considered acceptable.

Pollutant sources from residential subdivision were determined from a review of the literature since well-documented research on this topic has not been conducted in Maine. The applicable geology has been determined by examination of the literature, field mapping, aerial photo interpretation, auger drilling in the area, and interpretation of information gathered during a well survey that was conducted for this study. The ground water hydrology has been defined with a series of water level observations which were used in turn to calibrate computerized numerical aquifer simulation models. These computer models were used to model pollutant convection and dispersion in both the horizontal and vertical plane on both a regional and local scale. The criteria for determining whether or not the development impact is acceptable is taken from the EPA (1980) proposed "Ground Water Protection Strategy".

3.0 Pollutant Sources

Residential developments affect ground water quality in two main ways: a) through subsurface sewage disposal systems (leachfields), b) through lawn and garden fertilization and addition of manure from family pets. Recently, ground water quality surveys have begun to quantify the impact associated with residential developments, unfortunately, these studies rarely take into account the hydrologic regimes specific to each research area. There have been no detailed studies with reliable results reported for residential developments in Maine. Therefore, I must rely upon literature results from other areas of the world, which report a somewhat wide range in specific constituent concentrations of pollutants deriving from residential development.

3.1 Septic Tank Impacts

There is general agreement that nitrate-nitrogen (nitrate-N) is the limiting contaminant from residential development with onsite septic sewage disposal (e.g., see EPA, 1978, p. 466; Hook, et. al., 1978, p. 12; Rice and Raats, 1980). In other words, nitrate-N is the contaminant of most environmental and health concern. Nitrate-N has public health significance in that it can cause methemoglobinemia ("blue-baby syndrome") in infants when nitrates are present above certain concentrations in drinking water. There is also a suggested link between high nitrate levels and gastric cancer (World Water, 1980).

There is a wide range of estimates of the concentration of nitrate-N concentration that will appear in ground water below leachfields. In addition to variations in the nitrogen content of effluent entering individual leachfields, the concentration of nitrate-N entering ground water beneath the leachfields seems to depend upon soil texture, distance of unsaturated flow, and the ratio of carbon to nitrogen in the soil.

Bacteria and viruses are the other main constituents of septic tank effluent that are of environmental concern. Bacteria are attacked and consumed by organisms in the soil beneath a leachfield and viruses readily adsorb to soil particles and can be destroyed by soil microflora (Viraraghavan, 1980). Bacteria and viruses are subject to die-off and dilution with time; thus, the travel time of leachfield effluent to the nearest surface water body or well has been an important criterion to determine the separation distance of leachfields from wells and surface water. More research is needed on this aspect, but it is generally agreed by sanitary engineers that 100 to 300 feet of soil travel will provide adequate protection to a well from bacterial and virus contamination.

There are two main variables involved in determining the nitrate contaminant potential of leachfields: a) the nitrogen concentration of effluent leaving the sewage treatment system, and b) the nitrate-N concentration that reaches ground water immediately below a leachfield. Hook, et. al., 1978, cites statistics that found the average total-N concentration in septic tank effluent to be 55 mg/l. DeWalle and Schaff (1980) list the representative total-N concentration as 40 mg/l in their study on ground water pollution by septic tank drainfields. Olson, et. al. (1980) list the average total-N concentration in primary effluent at a rapid infiltration site as 40.2 mg/l with a range of 29.7 to 58.5 mg/l. Viraraghavan (1980) lists three studies where ammonia-nitrogen (the form of almost all nitrogen in septic tank effluent) ranged from 14 to 130.8 mg/l.

The combination of the negative ionic charge on soil particles and positive charge on the ammonium ion leads to nitrogen fixation. The ammonia-nitrogen is oxidized in the aerobic soil below a leachfield and converted to nitrate-N by nitrifying bacteria. Many of the mobile nitrate ions enter the ground water system and are reduced thereafter primarily only by dilution and dispersion. Organic materials in a leachfield (e.g., from garbage disposals) can be utilized by bacteria in conjunction with nitrates to denitrify the nitrified septic tank effluent and thereby create a significant reduction in nitrate-N additions to ground water. Some of the nitrate-N is also taken up by plant roots.

In several previous studies of this type, I have used 30 mg/l (see EPA, 1977, p. 467) as the appropriate value of nitrate-N reaching ground water. Kerfoot and Skinner (1979) found a breakthrough of total-N content from leachfields to be 3% to 49% with a mean of 16% of the typical effluent concentration. In a high density coastal town north of San Francisco that is served by individual septic tanks and leachfields (Wilson, et. al., 1979, p. 590), nitrate-N concentrations in shallow monitoring wells were less than 2.5 mg/l. Similar results were found in the Washington State study by DeWalle and Schaff (1980, p. 636).

In the latter three studies, specific correlations to leachfield densities, soil textures, and water table conditions were not given. In a more controlled study, however, of a Hollister, California, rapid infiltration site (Olson, et. al., 1980, p. 893), total-N in a sandy aquifer immediately below the site was only 7% of input levels. In the Hollister study, where the water table is relatively deep (20 to 30 feet below ground), the authors concluded that "denitrication represents an important mechanism for nitrogen removal..." and that "flooding frequency, soil pH, and carbon-to-nitrogen ratio all appeared favorable for the denitrification process." A high loading rate of 6310 kg/hectare of total-N was applied at Hollister. Hook, et. al. (1978, p. 12-15), in contrast, cite a number of cases where nitrate-N concentrations in ground water near leachfields were greater than the Safe Drinking Water Standard of 10 mg/l. The Hook, et. al. literature search fails to describe in detail all of the soil texture and ground water regimes that apply to the nitrate pollution examples that they cite. They briefly discuss the nitrate contamination by leachfields of a shallow aquifer in Nassau Co., Long Island, New York. Investigators concluded in that particular case that there was insufficient dilution in areas where the population was in excess of 3 persons per acre.

Taking all of the literature results into account, I conclude that the total-N leaving a leachfield will probably not average more than 60 mg/l, but that probably not more than 50% or this amount will enter the ground water and be converted to nitrate-N. Certainly an assumption that 40 mg/l of nitrate-N will be generated by each leachfield and reach ground water seems to be a conservative approach and is, therefore, adopted here.

3.2 Nitrogen sources other than septic systems

Fertilization of lawns and gardens will probably occur and some nitrogen may reach the ground water from this source. This is not a contaminant source that has received much attention and I do not know of any research results that enable me to quantify this contaminant source in a reliable manner. Repeated heavy applications of fertilizers over large areas, which are typical of agricultural practice, have been found to be associated with higher nitrate-N concentrations in ground water than that under residential areas. Yoder, et. al. (1980, p. 7) found mean nitrate-N levels of 2.28 mg/l in agricultural areas, compared with 0.43 mg/l in residential areas. World Water (1980) cites findings in chaulk aquifers in England that were contaminated with nitrate-N above acceptable levels by agricultural fertilization. Research showed that when nitrate and chloride were applied jointly to a crop in England, only 1% to 2% of the applied nitrate-N was found in lysimeters under the field compared with about 10% to 20% of the applied chloride, implying that the nitrogen tends to be tied up in the general biomass of the soil.

There is greater concentration fluctuation with time associated with intermittent land application of fertilizers than the essentially continuous subsurface applications of nitrogen from leachfields. Yoder, et. al., 1980, describes the peaks of nitrate-N caused by leaching deriving from agricultural land after heavy rains following dry periods. Rice and Raats (1980) describe the fluctuation and attentuation of nitrate-N peak concentrations associated with intermittent loadings of sewage on a local water table (e.g., from a rapid infiltration site). Yoder, et. al., 1980, describes the association of elevated levels of potassium with fertilization compared with the elevated levels of sodium that occur in association with septic tank discharge.

Several literature sources report nitrate-N concentrations in ground waters under sewered high density residential areas as being in the range of 1 to 2 mg/l and by inference one might imply that this represents the average potential nitrate-N contamination from sources other than leachfields. My interpretation of the water test results in Table 2, however, is that the nitrate-N concentrations correlate closely with probable leachfield solute sources and do not suggest large increments from "other sources", although significant portions of the aquifer recharge area are already developed at densities equivalent to the proposed Bay Park. I doubt that these other sources--such as fertilizers--are important in the modelled area.

The model results were derived from applying the contaminants in a "diffuse" manner. Any additional pollutant applications, such as fertilizer applications, would be modelled in the same way. Therefore, the distribution of any incremental contamination from non-point sources will have the identical distribution (although different concentration) as shown, for example, in Figs. 9-11.

The maximum incremental nitrate-N additions from "other sources" that I assume would be added in the study area is 1 mg/l, based upon my review of the literature and my experience in watershed studies for water utilities in other areas in Maine.

3.3 Applicable Impact Criteria

The Environmental Protection Agency (1980, p. 6) has issued a "Proposed Ground Water Protection Strategy" which states: "Until a classification system (for aquifers) is developed with full public participation and adopted, EPA will maintain a policy that where ground water is currently of drinking water quality or better, it will be provided protection to ensure that its utility for this use is not impaired." As stated in other portions of the EPA (1980) strategy, until better standards are developed, the Safe Drinking Water Standards will have to serve as the measure of acceptability for development impact. The conclusion at this point is that if a development will not cause an incremental degradation of drinking water quality in adjacent areas above the limits of the Safe Drinking Water Standards, the impact will be considered acceptable. The Safe Drinking Water Standard for nitrate-N is 10 mg/l.

4.0 Background Water Quality

In order to determine whether the incremental impact of the development on the ground water quality is within acceptable limits, the background quality must be known in the areas that would be affected by the development. The background quality has been measured on samples taken from monitoring wells and in the streams that drain the area. These streams (Black Alder Brook on the west and Foster Brook on the east) receive their flow largely from ground water discharge and therefore reflect ground water quality.

Table 2 summarizes the results of water quality tests obtained at the sampling points shown on Fig. 2. The highest nitrate-N reading was found at observation well A8 on 8/25/80, which is influenced by nearby septic systems in Phase I of Bay Park. Wells A4 and A12 are remote from any development and would thus represent true background levels of nitrate-N: about 0.1 mg/l. As one might expect from the results shown on Figs. 9 to 11, stream samples FB2 and BAB2 contained elevated levels of nitrate-N and bacterial contamination due to nearby development and possibly waterfowl activities in two ponds just upstream of the sampling points.

It appears safe to assume that the background concentration of nitrate-N in the ground water under and directly down-gradient of Bay Park is on the order of 0.1 mg/l without the superimposed effects of the Bay Park development. This is of similar magnitude to other Maine ground waters in undeveloped areas under forest cover.

5.0 Geology of the Bay Park Area

The basic geology of the area is described in Gerber, 1978. Smith (1977), Prescott (1968 and 1969), and U.S. Dept. of Agriculture (1970) all contain various interpretations of the surficial geology of the area. There is a southwest-trending bedrock trough under Bay Park that continues to the Androscoggin River and beyond. A test boring by the Brunswick-Topsham Water District (triangle shown on Fig. 2), which is mid-way along the Bay Park northern boundary, found 71 feet of sand over 76 feet of clay, over 2 feet of till with bedrock at 150 feet. To the northwest and southeast of Bay Park, bedrock ridges are exposed. A bedrock aquifer with high yield wells occurs in the bedrock ridge to the east, along Foreside Road. Small stratified moraines occur in the area of the Topsham landfill, northeast of Bay Park.

Auger drilling (Appendices I and II, Fig. 2) and calibration of the ground water model for this study indicates that the sand stratum becomes thinner on the southeastern corner of Stuart's land, and in the area between Foreside Road and the Androscoggin River. The auger holes, some of which were 51 feet deep, encountered predominantly stratified medium and fine sands overlying clay-silt deposits at variable depths. A thin (0.2') clay-silt stratum was found at 5 feet in hole A5; otherwise, the material was found to be all sand in all holes unless the bottom of the aquifer was encountered. At the bottom of the aquifer, the sand became very fine, then silty, and finally graded to soft silty clay. Grain size in all holes decreased from top to bottom generally, and is somewhat coarser in the north than in the south.

The Bay Park development is not in the direct recharge area of the bedrock aquifer to the east. The prime concern with ground water impact rests with the sandy aquifer between Bay Park and the Androscoggin River to the southwest. Therefore, this study deals with the dispersion of contaminants in this sandy aquifer, which serves many dug wells and well points along Foreside Road (Fig. 3 and Table 3).

6.0 Ground Water Hydrology

The ground water hydrology of the sandy aquifer of the Bay Park area has been investigated by field studies and detailed computerized simulation modelling. Seventeen auger holes were drilled in the aquifer to determine aquifer thickness, stratigraphy, and the potentiometric surface. Information on observation wells that were placed by Wright-Pierce-Barnes & Wyman at the Topsham landfill, north of Bay Park, was reviewed. Detailed topographic mapping and measurements of water levels are available for the land owned by the Town of Topsham, just north of Bay Park. All private wells in the aquifer down-gradient of Bay Park were surveyed (Table 3, Fig. 3) from which aquifer thickness and water elevations could be inferred. Information was obtained from Prescott (1967). The locations of streams that could act as full or partial "line sinks" were determined from U.S. Dept. of Agriculture (1970), through aerial photo interpretation, and by field mapping and survey.

6.1 Evaluation through Computer Simulation Models

The ground water hydrology of the Bay Park area was simulated in detail by developing numerical computerized ground water models of the aquifer. The ground water model was calibrated by drilling auger holes A1-A12 (Figs. 2 & 7) to ground water in and near Bay Park. The sandy aquifer is unconfined; therefore, the phreatic surface also represents the potentiometric surface of the aquifer with which we are concerned. The elevations of the water in each of the cased auger holes were accurately measured on the day after the holes were drilled. The locations of the holes and the elevations of the water in the holes as well as in all ponds, streams, and ditches were determined by survey. Elevations were converted to NVGD (USGS) datum by reference to the elevation of WPBW Test Well #2 (Fig. 2) at the Topsham landfill, which is referenced to NVGD. In some of the models, ponds, streams, and ditches--the ditches that will be converted to underdrains--that contained water in August 1980 were assumed to act as "line sinks" under conditions of normal precipitation recharge.

6.1.1 Water table elevations

The model was calibrated by simulating the ground water regime near Bay Park by varying transmissivities within the modelled area until an appropriate match is obtained with the measured water table elevations in the observation wells. Sensitivity analysis is provided to test the reasonableness of the model parameters. This type of approach to determining aquifer transmissivities over large areas is common and referred to as the "inverse problem". It is a valid approach if other variables and the boundary conditions are reasonably accurate. The alternative to back-calculating transmissivities is to conduct very expensive field drilling and pumping tests which can be difficult to interpret in low permeability materials such as fine sands.

Table 1 summarizes ground water elevations that were measured in the monitoring wells on 8/12/80, 11/26/80, and 2/26/81. Vandalism affected 6 of the wells so that readings could not be obtained on the latter two dates on those 6 wells. The latter two readings for well A9 appear suspect and may represent a change in the well rim elevation from which water levels were referenced. Observations seem to show that there is relatively little change in ground water elevations between the northern and southern boundaries of Bay Park and flow seems to be directed more toward the streams (line sinks) shown on Fig. 2 than from northeast to southwest. Average seasonal changes in the water table elevation are not great. The August readings were taken near the time of lowest water table, whereas the 2/26/81 readings were taken at the time of spring high water table. The reading in well A7 dropped after a ditch was excavated adjacent to it last fall. Well A4, which is nearest to the ground water divide in the north and would be expected to show the greatest elevation variation with the seasons, only increased about 2 feet from August to the time of snowmelt. Ground water tables lie at average depths of 4 to 6 feet below ground surface within Bay Park.

6.1.2 Transmissivities

Aquifer transmissivities, which were determined through the computer model calibrations, are shown on Figure 5. Transmissivity is the measure of ground water flow through a unit width of aquifer per unit time under unit gradient. It is a term that has meaning only when flow is relatively horizontal. Transmissivity is approximately equal to horizontal permeability times saturated aquifer thickness for relatively horizontal flow. "Apparent" transmissivities can be lower where gradients are high near streams and in anisotropic soils than would be the case for flow under low gradients. The vertically-averaged transmissivity within most of Bay Park is about 0.015 square feet per second, which equates to an average horizontal permeability of 18½ feet per day. This is the correct order of magnitude for the permeability of a fine to medium sand. Notice that the transmissivities are interpreted to be lower in the eastern and southern portions of the modelled area, due primarily to smaller aquifer thicknesses, finer soil grain size, and higher gradients than in northern Bay Park. The transmissivities in the southern portion of the 400'x400' grid element modeled area are not known with any certainty, since there were no calibration points in that area. However, this does not affect the evaluation of Bay Park impact.

Runs 7, 1, and 4 on Table 1 illustrate the sensitivity of the computed model elevations to change in transmissivity in the Bay Park area.

6.1.3 Anisotropy

The aquifer is assumed to be anisotropic in that the vertical permeability is only 10% of the value of the horizontal permeability (as a consequence of horizontal stratification of the alternating fine, medium, and coarse sand beds). This anisotropy ratio was obtained in pumping tests that I have analyzed in other glacial outwash aquifers in Maine. This anisotropy is reflected in my vertical section modelling (Fig. 13B). Although not reflected in the modelling, I infer from the auger hole logs that permeability may decrease with depth in the aquifer since grain size appears to decrease with depth.

7.0 Methods of Ground Water and Pollutant Dispersion Modelling

The ground water regime is simulated with a two-dimensional finite-difference computer model developed and documented by Konikow and Bredehoeft, 1980. This model has been used to simulate the impact of other large residential developments--particularly the subsurface sewage disposal area impact--in other areas of New England (Heeley, 1980). The model uses a rectangular grid to simulate the potentiometric surface in each grid element and computes the conservative (i.e., pollutants are not removed in the soil by adsorption, reaction, or cation exchange) dispersion of solute throughout the ground water regime. Density contrasts are not considered to be significant: pollutants occur in low concentrations and can be treated as "tracers". Before using the model to simulate dispersion of contaminants, the model must be calibrated so that it accurately reflects the real ground water flow regime. Major variables include transmissivities, aquifer thickness, precipitation recharge, longitudinal and transverse dispersivities, anisotropy ratio, and pollutant application rates and concentrations. The model uses the "method of characteristics" to simulate pollutant dispersion, which is recognized as an excellent state-of-the-art method. This method produces some numerical oscillation, however (successive approximations oscillate around a mean when the steady state is approached), which is illustrated between 20 and 30 years on Fig. 12A. When this occurs, I take the average value of successive oscillations.

The model incorporates its own internal measure of error. This error calculation is stated on the applicable figures of this report.

7.1 Boundary Conditions

I developed models at 3 scales for this study, in addition to developing a vertical cross section model. The near-field boundary conditions are dependent on the far-field boundary conditions. Therefore, the entire area between the northern boundary of Bay Park and the Androscoggin River south of Foreside Road was included in the model with the large grid size (400'x400'). This model was used to establish boundary conditions for the detailed model with 20'x40' grid element size.

Surveying and water level measurements for this study were combined with the information gathered by Wright-Pierce in their work on the Topsham landfill. It is clear that in the sandy aquifer there is a ground water divide that lies about 300 to 400 feet north of and paralleling Stuart's northern property line. The northern limit of the modelled area on Fig. 6 lies approximately on the ground water divide and therefore represents a valid "no-flow" boundary. The Androscoggin river at the southern end of the modelled area in Fig. 6 represents a valid constant head boundary. (A constant head boundary is a large body of relatively constant elevation, which controls the level of the local ground water table. A line sink is a narrow water body with constant water elevation at given points along its length and has the ability to remove ground water flow that enters the stream without changing the water elevations in the stream. True line sinks penetrate the full thickness of an aquifer.)

The streams shown as line sinks on Fig. 2 are only partial line sinks in the north, but nearly full line sinks in the south. Since the surface drainage within and near to the site that I model as "line sinks" are not fully-penetrating, they do not fulfill the theoretical requirements of "line sinks". Since these streams and drainages do flow from ground water recharge during all but severe drought conditions, however, it is clear that they have some importance to maintenance of local ground water tables. This conclusion is reinforced by modelling studies on the Kennebunk outwash aquifer where I found that the many small individual streams are very important in controlling local water table elevations. The particular computer model that I used for the Bay Park study does not have the capability to incorporate partially-penetrating streams on a regional scale; therefore, I have modelled the pollutant dispersion both with and without the inclusion of line sinks within and near the development.

Although the no-flow boundaries of Fig. 2 to the east and west are technically not always on the drainage divides, this does not affect the modelling of dispersion within Bay Park, since line sinks are specified between those areas and the area of impact.

The intermediate scale model (250'x250' grid element size, Table 1), which was discussed in more detail in Gerber (1980), does not have a valid no-flow boundary on the southern edge. The effect on model results is only prominent near the southern edge and the use of the 400'x400' grid element size model overcomes this problem on a regional scale.

The detailed models (40'x20' and 40'x4' grid element size, Figs. 13A & 13B) are generated by placing linear line sinks at each end of the model with the difference in head between the line sinks equal to the differences obtained from the ground water contours of the far-field model for the same relative locations.

7.2 Transmissivity

After establishing the grid, boundary conditions, and precipitation recharge, trial transmissivities are assumed in each grid element and the model predictions of water table elevation are compared with the 12 observation well readings and inferences of water table elevations in other parts of the model ("history matching"). These calibrations are made with water tables generated under "steady state" conditions. Transmissivities are adjusted until the differences are within acceptable range (about 1 foot for this application). Primary transmissivities of 0.02, 0.015, and 0.007 square feet per second within the majority of Bay Park were tried under conditions of average recharge. The transmissivity array that provided the best match is shown on Fig. 5.

Notice that a very close match was obtained (Table 1) between observed water table elevations and predicted elevations (Run 36 and Run 4). The predicted elevation should be somewhat higher than those observed in August, since the August levels should have been below the "average" position of the water table.

Table 1 summarizes a sensitivity analysis on the primary transmissivity assumption with the 250'x250' grid element model (Runs 7, 1, and 4). Because of the boundary conditions on the southern edge of the model (just south of Bay Park's southern boundary), simulation results for A11 and A12 overestimate actual values. Notice that for the other observation wells in this model, which includes line sinks within the developments, Run 4 provides a very good fit with the existing situation.

7.3 Aquifer Thickness and Porosity

The Brunswick-Topsham Water District boring at the northern edge of Stuart's property found a 71-foot thickness of sand overlying clay. Fifty-one foot auger holes (Appendices I and II, Fig. 2) were drilled at points throughout the aquifer. Fig. 4 presents the contoured thickness matrix derived from geologic studies. This thickness matrix was used in the dispersion modelling.

The value that is assumed for aquifer thickness makes no difference in calibrating the transmissivities of the model to produce an accurate representation of the water table position. The thickness only affects: a) the time required for establishing a steady state pollutant distribution, and b) the degree of dispersion, since dispersion is proportional to seepage velocity which is equal to transmissivity divided by aquifer thickness divided by porosity. Porosity is assumed to by 35% based on porosity tests made in the Kennebunk outwash aquifer as reported in SEA Consultants, Inc., 1979.

The thickness matrix portrayed on Fig. 4 may overstate the thickness in portions of the aquifer; however, this leads to conservative results in the pollutant dispersion calculations. Since dispersion is proportional to velocity and velocity is proportional to transmissivity divided by thickness, overstating thickness underestimates seepage velocity which overestimates pollutant concentrations.

7.4 Precipitation Recharge and Pollutant Application

7.4.1 Precipitation Recharge

Since the northern edge of Bay Park is near a ground water divide, it is obvious that the natural factor creating and sustaining the local water table is precipitation recharge, which averages about 44 inches per year in Topsham, but has been as high as 60 and as low as 28 inches per year. Although precipitation is not distributed uniformly over the year, but is a stochastic process, it is common practice to treat the recharge as uniform in long-term simulations, due to the slowness of ground water movement as well as the difficulties of simulating stochastic processes. Watershed yields in Maine average 55% to 65% of incipient precipitation and I have found that the recharge reaching similar sandy aquifers in southwestern Maine is about 60% of precipitation when averaged over time. Since runoff is negligible for the sandy aquifer under and near Bay Park in its undeveloped state, almost all precipitation that is not evaporated or transpired goes to ground water recharge which in turn is the source of water for the nearby streams on a year-round basis. Thus, in its natural state, I assume that the aquifer is recharged at the rate of 60% of annual precipitation, which equates to average rates of 4.4×10^{-8} feet/second for drought years and 7×10^{-8} feet per second for average annual conditions. Since it takes 30 to 40 years for steady state contaminant distribution to occur in Bay Park alone (Figs. 12A and 12B), one can see why use of average annual recharge rates are the appropriate choice for long-term simulation.

Table 1 summarizes the results of water table elevation sensitivity to choice of recharge rate. Comparison of Run 35 and Run 36 show that a decrease of about 15% of recharge over the 400'x400' grid element model results in an average decrease in steady state water table of about one-half foot. Comparison of Runs 4, 5 and 6 (250'x250' grid element model) suggest that the water table will range 3 to 5 feet in the north and up to 8 feet near Stuart's southern boundary between a drought condition and spring high water table conditions.

7.4.2 Pollutant Applications

Recharge from individual leachfields is added in the contaminant dispersion analysis, since water is piped into the houses by the local Water District. Although the addition of storm sewer will cause some runoff and thus a loss of recharge compared with the present condition, Water District records show that the average residential customer will put about 33 cubic feet per day into their leachfields, which will probably about offset the loss of recharge from the development. The leachfield recharge rate amounts to about 1.5×10^{-8} feet/second when applied uniformly over a developed area with a density of 0.6 acres per lot. This is about 20% of the present average recharge rate. My computer analysis implies that this would cause a rise in the developed area of about ¼ to 3/4-foot in the average position of the water table, without taking into account the loss of recharge from the addition of impervious area. Taken in the balance, I would expect these two effects to cancel each other out. Computer runs from which Figs. 13A and !3B are taken showed that the leachfield recharge spreads out rapidly through the aquifer, rather than producing a pronounced mound near the leachfield. The mound near the leachfield will only be a maximum of 0.15 foot higher than the adjacent water table.

To reflect a loss of precipitation recharge from addition of impervious areas as part of residential areas (typically 15% impervious area), I have reduced the rate of average recharge 15% (to 6×10^{-8} feet/second) for Bay Park and similar built-up areas within the modelled area. Depending on existing or planned densities in each cell of the model, the effective leachfield recharge rate for that cell is added to 6×10^{-8} feet/second. The ratio of leachfield recharge rate to total (leachfield + precipitation) recharge is an averaged input concentration as a percent of leachfield contaminant concentration. By estimating the input concentrations, the absolute value of the ground water concentration can be calculated at any point in the modelled areas. Vertical section models provide a perspective on the vertical distribution of the contamination.

For the far-field analysis, which necessitates a large grid element size, the contaminant distribution is somewhat generalized. However, I have generated detailed models with grid elements (20'x40') that approximate the size of a single leachfield (Fig. 13A) which illustrates the near-field distribution of contaminants for two rows of aligned leachfields. In all cases, the precipitation and pollutant recharge are treated as diffuse sources rather than point sources (such as injection wells). The model thus applies the pollutants uniformly over the source cell area. This method is also applicable to evaluating the effects of area-wide lawn fertilization, etc.

7.5 Dispersivity

Conservative (non-reactive) pollutants are transported and diffused through the aquifer by the process of convection and dispersion. Convection results solely in dilution along the ground water flow channel in which the pollutant enters. As a fixed mass of pollutant solute passes farther and farther along a flow channel, it is mixed with more and more water volume, reducing its concentration with distance from a source. Dispersion, on the other hand, is a mechanical mixing process that operates on a molecular level as well as a macroscopic pore-water level to cause a spreading of the contaminant along the path of travel as well as perpendicular to it (across flow lines). "Hydrodynamic dispersion" is an important process in reducing point concentrations of contaminants in the near-field in sandy aquifers. Dispersivity is an intrinsic property of an aquifer with a given scale and geometry. In the theoretical pollutant transport equation (p. 3 of Konikow and Bredehoeft, 1978), the dispersion of pollutants is proportional to dispersivity and to aquifer seepage velocity vectors (Scheidegger's equation). Change in concentration due to dispersion is proportional to concentration gradient at a given point. In the mathematical simulation of dispersion, some dispersion may be predicted up-gradient of the source, whereas this is not observed in nature.

No calculations of dispersivity derived from field measurements have been made in this type of aquifer in Maine; however, it is clear from the qualitative results of unpublished studies that dispersion is a real and important process. The dye tracer tests in the Kennebunk outwash aquifer by SEA Consultants, Inc. (1979) showed the dye fanning out at a minimum 45^{o} angle from the point sources (transverse dispersion) and longitudinal dispersion accounted for the dye reaching down-gradient wells far in advance of the time that would be predicted by the soil permeability.

I have used values for longitudinal and transverse dispersivity of 30 feet and 10 feet, respectively, which are conservative when compared with values used by Swain and Pinder (1977, p. 305) for an alluvial aquifer in California. Values from Fried (1975, e.g., p. 305) are of the same order of magnitude for aquifers having approximately the same transmissivity as at Bay Park. Faust and Mercer (1980, p. 571) cite values of longitudinal and transverse dispersivity obtained from a thorough literature review as 21 meters and 4 meters, respectively, for "glacial deposits", and 12-61 meters and 4-30 meters for alluvial deposits. Therefore, I feel that the values I selected are appropriate and probably conservative. As discussed in Swain and Pinder (1977), the solute concentration distribution is not very sensitive to great variations in assumed dispersivity in the far-field analysis.

7.6 Simulation Time

The desired result of this ground water modelling effort is to be able to predict the maximum concentration of the most critical pollutant in the ground water as it leaves the boundaries of the Bay Park development. It is important to determine how much simulation time is necessary to reach the "steady-state" contamination distribution. As noted earlier, simulations have been made under the alternative assumptions that line sinks both do and do not occur within the Bay Park development. The presence of line sinks or even partially-penetrating streams acts to remove ground water locally, keeping the overall north-to-south ground water gradient rather small. This increases the required time to reach a steady-state condition for the case where the line sinks are assumed to be present.

Figs. 12A and 12B are plots of concentration change with time at grid element (7,7) of Figs. 10 and 9, respectively. The hydrodynamic dispersion causes the curves to be "S-shaped". Once the steep portion of the curve has flattened, it will begin to approach an asymptote that is equivalent to the steady state condition. Fig. 12A shows that it takes approximately 28 years for contaminant distributions to reach steady state within Bay Park under the assumption that no line sinks are operative within the development. Fig. 12B, which illustrates a more diffuse flattening of the concentration curve with time, takes something on the order of 40 years to approach steady state conditions if line sinks are operative in Bay Park. The detailed simulations that are shown on Figs. 13A and 13B represent a total simulation time of 28 years, which are approximate steady state conditions for the respective model assumptions.

8.0 Discussion of Computer Modelling Results

The following sections discussion the results and implications of the model simulations that I made as part of this study.

8.1 Ground water Flow

Figures 6 through 13, inclusive, summarize the important computer results from over 50 separate model runs. Fig. 6 shows far-field model boundaries; Fig. 7 (and 2) shows the location of the calibration points. The line sinks that are applicable to each model result are shown on the respective figures.

8.1.1 Figure 6

Figure 6 shows the predicted ground water contours (NVGD datum) for the steady state condition once Bay Park is developed in conjunction with existing development. Accuracy may be low in the southwest where no data are available for model calibration; however, this does not affect the results of this dispersion study. Agreement with actual data is good in the area of Bay Park.

8.1.2 Figure 7

Figure 7 shows hypothetical steady state ground water contours for the modelled area under the assumption that no line sinks occur in Bay Park. The water table contours are obviously not in accord with the actual field conditions, which implies primarily that the line sinks are important to the ground water regime of Bay Park and secondarily that the transmissivities (Fig. 5) that were used to generate Fig. 7 may be lower than actual. As I previously described, however, the latter case does not affect aquifer flux or seepage velocities for the assumption that no line sinks occur in the development.

8.1.3 Figure 8

Figure 8 represents a transient ground water contour condition beginning with the contours in Fig. 6, then removing the line sinks in the Bay Park area and reducing precipitation recharge to that of a severe drought condition. The ground water contours are allowed to adjust to the hypothetical condition that would occur two years hence under these conditions. Note that the predicted elevations are higher than at present, which implies that a) at least part of the line sinks are important during part of the drought, b) the values in the transmissivity matrix may be too low in the area of Bay Park. Refer to Run 6 on Table 1 (the transmissivities along Foster Brook are lower than for Run 36, which increases the predicted level at A10 over Run 36). Notice that even in a "steady state" drought condition, the ground water table will still be higher than, and therefore flow toward, the adjacent line sinks. It appears, therefore, that although the transmissivities may be somewhat understated, the line sinks are still important during transient droughts.

As with the case of Fig. 7, the flux and seepage velocities are correct for the case of drought and no line sinks operative in Bay Park.

8.1.4 Figures 13A and 13B

The equipotential lines for Figs. 13A and 13B are not reproduced in this report; however, it is important to note some basic aspects of the flow regime that applies to these figures. Figs. 13A and 13B were developed from the ground water flow regime represented in Fig. 7. In Gerber (1980), I had shown how the inclusion of theoretical line sinks in Bay Park resulted in the local removal of pollutants from the ground water into the surface water, thus reducing the mass of contaminants that would pass in ground water flow through Bay Park's southern boundary. Due to the difficulty in modelling the three-dimensional effects near the partially-penetrating streams in Bay Park, a conservative approach was selected to model pollutant transport through Bay Park's southern boundary.

The northeast-to-southwest trend through the middle of Figs. 9 and 10 (on a line through element 7,7 at points of highest concentration along the southern boundary of Bay Park) was chosen as the assumed path of maximum pollutant transport in which line sinks were not available to remove pollutants from the ground water flow. It follows from the basic flow and conservation of mass equations that if a ground water divide occurs to the north of Bay Park and the true line sinks lie to the south of Bay Park, that flow must be perpendicular to the contour lines on Fig. 7 and the product of the gradient times transmissivity must equal flux (recharge) applied to the aquifer along the flow path. Even though the gradient may be wrong as calculated from Fig. 7, the gradient in combination with the transmissivity, aquifer thickness, and porosity will yield the correct flux and seepage velocity along the flow path.

The gradients, transmissivities, and aquifer thickness of Fig. 7 are reproduced in the small grid size element model of Figs. 13A and 13B to represent the case of no line sinks operative in Bay Park. Programming constraints required that the total length of the model involve three separate model simulations, beginning at the north side of Bay Park. The approximate layout of leachfields corresponds to the Bay Park subdivision plan. For pollutant transport, the steady state distribution of contaminants leaving the down-gradient side of one stage of the model is the source input for the next down-gradient stage of the model. The vertical cross section model is anisotropic and a detailed flow net shows the equipotential lines curving up-gradient slightly as depth increases along the equipotential line.

8.2 Pollutant Dispersion

Given a ground water flow regime, the models track the movement and concentration of contaminants throughout the aquifer. Figs. 9, 10, and 11 represent the distribution of contaminants on a regional scale for the respective flow fields in Figs. 6, 7, and 8. To take existing known sources of leachfield contaminants into account in the same manner that I handle the Bay Park development, I have used tax maps and recent aerial photographs to locate existing houses that contribute to the modelled area. The contributions from the houses along Rt. 24 and Foreside Road that were located in or draining into the modeled area are added either at the boundaries of the model or at the actual location of the houses if they were located within the modelled area. The effects of the subdivision in the southwestern corner of the modelled area (e.g., Fig. 9), which is approximately the same density as Bay Park, are similar in magnitude to Bay Park (vertically-averaged pollutant concentrations are about the same).

The values of the isocon contours in Figs. 9, 10, 11, and 13A are the vertically-averaged concentrations of leachfield inputs as a percentage of the input concentration. Isocons are lines of equal chemical concentration. Since nitrate-N is the contaminant of concern and it input concentration is assumed to be 40 mg/l, multiplication of 40 mg/l by the percentages on the isocon lines yields the ultimate concentrations. A difficulty of the regional scale model is that the diffuse contaminant sources must be distributed uniformly over a cell so that it is not possible to illustrate the detailed variations in contaminant distribution within Bay Park as the small scale model such as Fig. 13A could show. As discussed below, the dispersion of contaminants is great such that south of Bay Park local pockets of above-average concentration disappear.

8.2.1 Figure 9

Figure 9, the case with line sinks in Bay Park, represents the approximate steady state contaminant distribution at Stuart's boundary after 40 years of flow under average recharge conditions. The change of vertically-averaged concentration with time at Stuart's boundary (element 7,7) is shown in Fig. 12B. The predicted nitrate-N concentration at the boundary is 4 to 5 mg/l from the leachfield sources.

8.2.2 Figure 10

Figure 10, the case with no line sinks assumed in the development, predicts that the vertically-averaged nitrate-N concentration is about 14½% of input concentration or 5.8 mg/l (see Fig. 12A). Therefore, more pollutant mass is transported across Bay Park's southern boundary in ground water than in the case where line sinks are assumed to be operative in the development.

8.2.3 Figure 11

Figure 11 is a hybrid simulation employing a 2-year transient simulation of ground water flow without line sinks in Bay Park but beginning with the realistic water table conditions of Fig. 6. However, the solute distribution of Fig. 10 is used as the starting contaminant distribution. There is no noticeable change in the vertically-averaged concentration at Bay Park's boundary (e.g. at element 7,7) within the limits of accuracy of the model. This is not surprising considering the long period of time required for pollutant distribution in the aquifer. There is a tremendous volume of water under Bay Park, which makes the incremental mass of contaminants added during a one or two year drought appear rather insignificant.

8.2.4 Figures 13A and 13B

Figures 13A and 13B summarizes a complex (and expensive) 28-year simulation of the detailed near-field distribution of leachfield contaminants that would occur for the case where two parallel lines of eight leachfields occur in a flow regime from northeast to southwest across Bay Park without intervening line sinks. The leachfield locations are shown in plan view in Fig. 13A. Notice that the highest localized concentrations of vertically-averaged contaminants are in the vicinity of the third leachfield from the right. Notice how hydrodynamic dispersion causes contaminants to spread out away from the leachfield sources and how concentrations between leachfields steadily increase until about 300 feet down-gradient of the last leachfield (on the left) where concentrations begin to decrease.

Figures 13A and 13B have several special anomalies due to the methods that had to be used to prepare them. First, because of the large dispersion that occurs, there will be additions of pollutants to the water columns represented in Fig. 13A from leachfields to the sides of the two lines shown. Thus the vertically-averaged concentrations down-gradient of about the second leachfield from the right on Fig. 13A will be somewhat understated due to the additions due to transverse dispersion from leachfields to either side. We see from Figs. 10 and 12A that the vertically-averaged concentration at Stuart's boundary is about 14½%, whereas on Fig. 13A it is shown as approximately 10% of input concentration. This discrepancy will be taken into account below.

Figure 13B represents the vertical distribution of contaminants under one of the lines of 8 leachfields. The modelling stability constraints required an extremely high number of time steps (and very high computer costs) if hydrodynamic dispersion were included in the vertical plane model. Therefore, the contaminant distribution in the vertical cross section model was simulated as a convective flow only, with no dispersivity. This results in a conservative and unrealistic situation without the benefit of longitudinal dispersion in the direction of flow, the transverse dispersion in the vertical direction, and the transverse dispersion in the plane normal to the vertical plane. Using the results of the plan-view vertically-averaged models, it is possible to correct for the dispersion in the plane transverse to the vertical plane and in part for the longitudinal dispersion. The percentage concentration of contaminants in each cell of the convective vertical plane model was multiplied by a ratio that represents the vertically-averaged mass concentration along the section line in Fig. 13A to the vertically-averaged mass concentration in the respective column of the vertical cross section. This process is valid except for the immediate vicinity of a leachfield (source) where the high localized addition of contaminants in only the top portion of the aquifer causes an uneven weighting to occur in these columns. For example, the average ratio of dispersed to undispersed vertically-averaged mass concentrations in columns not immediately under or adjacent to leachfields is about 30%, but can reach 45% under leachfields. This created the increased localized concentrations in the depressed contaminant plumes in the vertical section of Fig. 13B. This effect would not be observed in the field. Conversely, in the upper portion of the section, immediately under the leachfields, concentrations should be somewhat higher than shown on Fig. 13B. Results of this calculation process should be approximately correct at distances of more than 2 cell lengths (80') from a leachfield location.

The vertical section (Fig. 13B) suggests that the peak steady state concentration of leachfield contaminants at the Bay Park boundary is 15% of input concentration, or about 6 mg/l of nitrate-N. However, as described above, this does not include the incremental amounts from transverse dispersion of more distant leachfields as discussed above. Using the same ratio and multiplicative process between Figs. 10 and the data base for 13B, the highest percentage concentration in the vertical section at the Bay Park boundary is 22% or 8.8 mg/l, occurring in the same relative locations as shown in Fig. 13B. When added to the 0.1 mg/l background level, and a hypothetical incremental 1 mg/l for point sources, we arrive at a total maximum hypothetical concentration of 9.9 mg/l which is just under the Safe Drinking Water Standard of 10 mg/l. I emphasize, however, that this conclusion is built upon multiple conservatisms.

9.0 Summary and Conclusions

1. This report estimates the maximum concentration in ground water of the pollutant of maximum concern arising from Bay Park Phases I, II, III and IV, which is or is proposed to be built with onsite septic tanks and leachfields. Pollutant source concentrations are chosen as conservative averages from a search of current research results in the United States. Nitrate-nitrogen is the contaminant of most concern and will reach ground water immediately beneath leachfields in concentrations averaging 40 mg/l. Incremental sources such as lawn fertilization and pet manure will add an incremental 1 mg/l over the entire Bay Park area. If the concentration of nitrate-nitrogen at any point on Stuart's Bay Park boundary is less than 10 mg/l, the impact is considered acceptable under the EPA's "Proposed Ground Water Protection Strategy".

2. Bay Park is at the head of a sandy unconfined aquifer. The extent, thickness, and water table within this aquifer has been investigated by field drilling and surveying. Boundary conditions on the aquifer have been determined by field mapping, survey, aerial photo interpretation, and reference to geologic literature sources. The private water wells down-gradient of Bay Park were surveyed as part of this study, and include numerous dug wells and well points. Background concentrations of nitrate-nitrogen in undeveloped areas in and near Bay Park were measured in wells and streams and found to be 0.1 mg/l.

3. The transmissivities within the aquifer and associated ground water flow regime were determined with numerical computerized ground water simulation models. Models were derived at several scales and included variable assumptions concerning the effectiveness of streams within and adjacent to Bay Park in directing ground water flow and removing pollutants from the ground water. Pollutant transport and dispersion was modelled in both horizontal and vertical planes and included the effects of existing developments as well as all of Bay Park.

The models show that it will take on the order of 30 to 40 years to develop a steady state contaminant distribution within Bay Park, following the time it is fully developed.

4. The following conservatisms are included in the modelling:

a) nitrate-nitrogen concentrations reaching ground water beneath leachfields are chosen to be higher than that used in the majority of the literature, including an EPA document (1977) which used 30 mg/l instead of the 40 mg/l used here

b) transmissivities are conservatively utilized as lower than the probable field values which leads to lower permeability and thus lower seepage velocities with resultant lower dispersion under a given thickness and gradient

c) values of aquifer thickness are conservatively chosen as greater than the probable field values, which leads to lower seepage velocities and less dispersion

d) dispersivity values are chosen to be on the low side of values cited in the literature

e) in modelling the distribution of contaminants in the vertical plane, no transverse dispersivity is assumed. This leads to greater concentrations of pollutants in the vertical plane than would otherwise occur.

5. The results of the modelling predict a total nitrate-nitrogen concentration in ground water leaving Bay Park's boundaries from all sources of from 6 to 7.8 mg/l as a vertically-averaged concentration. At the single point where the concentration is conceived to be at the highest value under the conservative assumption that streams within and near Bay Park do not remove any contaminants, the total concentration is predicted to be 9.9 mg/l.

6. The impact of Bay Park on ground water quality is predicted to be acceptable under all normally foreseeable conditions and should generally provide a comfortable margin between the predicted levels and the Safe Drinking Water Standards. Under conservative assumptions, certain points along Bay Park's southern boundary may approach very close to the Standard. Wells along Foreside Road should see no more than an incremental 2 mg/l increase from Bay Park. No respondents to the well survey indicated any elevated levels of nitrate-nitrogen in their well water. Nearby stream levels are showing less than 1 mg/l.

7. The modelling suggests that there will be no decrease in ground water availability to down-gradient users as a result of Bay Park. Due to the rela-

tively low transmissivities of the aquifer, water table elevations in the aquifer are relatively insensitive to changes in recharge that might be associated with this development, even if it is ultimately connected to an offsite sewage treatment system.

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TABLE 1--Observation Well Data and Comparison with Computer-simulated Data

Obs. Well #	Grnd. Elev. USGS	Water Elev. 8/12/80 USGS	Water Elev. 11/26/80 USGS	Water Elev. 2/26/81 USGS	400'x400' Grid Water El Run 35	Water El Run 36	250'x250' Grid Element Model Simulation Water El Run 7	Water El Run 1	Water El Run 4	Water El Run 6	Water El Run 5
A1	66.0	59.05	59.50-	59.62-	59.30	59.56	61.45	58.54	59.16	58.35	61.04
A2	65.9	61.64	60.48-	60.48-	61.23	61.70	64.01	59.98	60.92	59.83	63.45
A3	66.0	61.75	61.93-	61.52-	61.86	62.46	67.26	61.02	62.67	61.05	66.45
A4	65.3	61.26	62.51	63.14	61.29	61.76	65.26	60.77	61.96	60.80	64.68
A5	63.7	59.73	59.66	60.23	59.93	60.29	59.30*	59.30*	59.30*	59.30*	59.30*
A6	65.7	59.22	59.60	59.17	61.08	61.62	66.18	59.95	62.05	60.62	65.37
A7	65.5	61.57	60.13-	60.13-	61.29	61.78	63.14	59.94	60.79	59.98	62.70
A8	67.0	58.73	58.76	58.76	58.49	58.58	60.49	58.47	58.92	58.37	60.18
A9	63.9	59.69	56.30?	55.75?	56.71	57.73	59.88	57.11	57.95	57.27	59.54
A10	63.8	57.47	----	61.37-	58.02	58.23	65.96	55.03	61.06	59.36	65.03
A11	65.1	59.04	----	60.80-	57.90	58.69	71.99	57.70	64.01	61.17	70.65
A12	66.3	58.38	57.14	57.29	56.53	57.36	68.87	58.84	62.52	60.24	67.81

Run #	Precipitation Recharge Rate	Primary Transmissivity	Comments
35	6×10^{-8} ft/sec	0.015 ft^2/sec	Calibration run to check precipitation sensitivity for present situation
36	7×10^{-8} ft/sec	0.015 ft^2/sec	" " " " " " " " "
7	7×10^{-8} ft/sec	0.007 ft^2/sec	" " " " transmissivity " " " "
1	7×10^{-8} ft/sec	0.020 ft^2/sec	" " " " " " " " "
4	7×10^{-8} ft/sec	0.015 ft^2/sec	" " " " " " " " "
6	4×10^{-8} ft/sec	0.015 ft^2/sec	" " " " precipitation " " " "
5	14×10^{-8} ft/sec	0.015 ft^2/sec	" " " " " " " " "

*Defined as being on a line sink

Notes:

1. Water elevations are referenced to WPBW Test Well #2 (Fig. 2) which has assumed USGS elev. 64.50' (MSL)
2. Water elevations on 11/26/81 and 2/26/81 given as "-" if sand or other obstruction found in hole without water (elevation noted is level of obstruction); water elevations not given ("----") if hole was completely plugged with sticks, etc.
3. With Runs 4, 5, 6, 35, & 36 different transmissivities were used in the southeast and/or southwest corners of the Stuart development

TABLE 2--Background Water Quality

Water quality tests by McFarland Assoc., Inc., on dates as noted[1]

	Locations						
	A4[5]	A8[5]	A12[5]	BAB1[6]	BAB2[6]	FB1[6]	FB2[6]
Fecal Coli.							
12/22/80[2]	0	0	0	0	0	0	0
2/25/81[3]	0	0	0	0	41	3	23
Nitrate-nitrogen[4]							
12/22/80	<0.01	0.80[7]	0.02	0.38	0.50	0.02	0.21
2/25/81	0.11	0.26	0.08	0.23	0.60	0.07	0.27

Notes:

1 see Fig. 2 for location of sampling

2 colonies per 10 ml

3 colonies per 100 ml

4 mg/l

5 taken in observation well near top of water table

6 surface water samples taken from mid-stream

7 8/25/80 test on A8 found 2.0 mg/l

TABLE 3--Summary of Well Data obtained from Survey along Foreside Road

Well No.*	Tax Map Lot No.**	Type of Well	Depth (feet)	Yield (gpm)	Year of first use	Static Water Level (feet)	Date Static level measured	Year well ran dry	Quality Tested?	Quality Good?	Comments
1	3	point	16	--	1940	12	11-66	Never	yes	yes	located near many springs
2	4	drilled	298	2	1978	150***	9-78	"	no	--	
3	4A	drilled	172	6	1978	50	8-78	"	yes	yes	some iron in water
4	6	point	--	--	1973	--	--	"	--	--	
5	7	dug	18	--	1965	14	12-80	"	yes	yes	
6	8	point	12-14	--	1952	--	--	"	yes	yes	
7	9	dug	--	--	1960	--	--	"	--	--	also has artesian well which is not used
8	10	drilled	248	2	1977	12	'79	"	--	--	water has sediment; also has dug well that went dry and 400' drilled well that pumped "muddy water"
9	10A	drilled	307	50-60	1975	3	'75	"	yes	yes	some iron in water
10	10B	dug	15-19	--	--	--	--	"	yes	yes	used more than 3 years; occasional "sulphur odor"
11	11	drilled	373	3	1978	40	fall '78	"	yes	yes	
12	12	drilled	190	--	1956	--	--	"	yes	yes	
13	13	drilled	254	2-2/3	1966	175***	5-66	"	--	--	
14	14	point	12-14	--	1950	--	--	"	yes	yes	
15	15	point	14	--	1965	--	--	"	no	--	
16	15A	point	15	--	1968	--	--	"	yes	yes	
17	16A	point	12	--	--	--	--	"	no	--	
18	20&22	spring	4-5	--	1920	2	12-80	"	yes	yes	some chloride in water
19	20A	drilled	225	--	1971	50	12-80	"	yes	yes	pumped dry while watering lawn during summer 1980
20	21A	drilled	265	30	1978	--	--	"	no	--	leaves residue on cooking utensils; oily taste?
21	23	dug	8	3	1979	4	12-80	"	yes	yes	originally had "iron fungus"
22	24	dug	6	--	1950's	2	12-80	"	no	--	
23	26	dug	19	--	1800's	16	12-80	"	no	--	
24	26A	dug	7	--	1979	1	8-79	"	yes	no	originally showed coliforms
25	26B	dug	18	--	1972	--	--	1975	yes	yes	"nearby road construction caused loss of water to well"
25	26B	drilled	190	35	1976	100***	10-76	Never	yes	yes	water has "sulphur odor"
26	26-1	drilled	85	18	1978	--	--	--	yes	no	treated to remove iron
27	26-2	drilled	223	50+	1980	10	7-80	Never	yes	no	treated to remove iron

TABLE 3 (continued)

Well No.*	Tax Map Lot No.**	Type of Well	Depth (feet)	Yield (gpm)	Year of first use	Static Water Level (feet)	Date Static level measured	Year well ran dry	Quality Tested?	Quality Good?	Comments
28	26-3	drilled	116	20	1978	20	5-78	Never	yes	no	treated to remove iron
29	26-4	drilled	123	10	1978	20	8-78	"	--	--	
30	26-5	drilled	220	50	1979	--	--	"	--	--	some iron in water
31	27	springs	4	--	1940	1½	12-80	"	yes	yes	2 springs used
32	27A	drilled	180	--	1976	--	--	"	yes	no	treated to remove iron and manganese; slight "sulphur odor"
33	27B	dug	8	--	1969	3	'79	"	no	--	water once had "odor"
34	27C	drilled	130	9	1969	32	'69	"	yes	yes	
35	27D	dug	--	--	1973	--	--	"	yes	yes	
36	27E	dug	30	--	1977	26	'77	1977	yes	yes	used spring (well #31) after water level dropped in dug well
37	30	drilled	200	--	1972	--	--	never	no	--	well produces "black sediment" after prolonged pumping; also has dug well

* See Locations on Figure 3

** Tax map lot number on Topsham property tax map R-8

*** these great static levels (measured below ground surface) which were measured at the time of drilling probably represent the depth at which the water vein was encountered and not necessarily the present static level which is probably much closer to ground surface

Note: "--" indicates that the survey respondent did not know this data

APPENDIX I--Auger Hole Logs (see Fig. 2 for locations)

by Edward Bradley, Certified Geologist

A1

0- 2.0'	*sandy brown loam (topsoil and/or part artificial fill)*
2.0- 7.0'	*medium to coarse sand, some fine sand; tan to brown*
7.0- 9.0	*medium to coarse sand, scattered fine gravel with a few pebbles*

A2

0- 1.0'	*dark brown silty loam topsoil*
1.0- 7.5	*medium to coarse sand, some fine light brown sand; partially sorted*
7.5- 9.0	*fine to coarse sand; buff to light gray*

A3

0- 1.0'	*brown sandy loam topsoil*
1.0- 2.5	*fine to medium sand; rusty brown*
2.5- 8.5	*fine to medium sand; light gray to buff; very micaceous*

A4

0- 1.0'	*brown sandy loam topsoil*
1.0- 2.5	*very fine to medium sand; rusty brown; micaceous*
2.5- 9.5	*very fine to medium sand; buff to light gray*

A5

0- 5.0'	*medium sand with some fine sand and some coarse sand; buff at top, buff to light gray at 2.5'-5'*
5.0- 5.2'	*silt and clay; bluish-gray with brown staining in vertical joints*
5.2- 8.5	*very fine to medium sand with a little silt; buff to light gray*

A6

0- 0.3'	*brown sandy loam topsoil*
0.3- 1.0	*fine to medium sand; rusty brown; somewhat micaceous*
1.0- 9.0	*fine to medium sand; light gray to buff; less micaceous near bottom*

A7

0- 0.3'	*very sandy brown loam topsoil*
0.3- 1.5	*fine to coarse sand; rusty brown*
1.5- 5.5	*fine to coarse sand; buff to tan*
5.5- 6.0	*fine to medium sand; buff to tan; "clay balls" 1"± diameter: silt and clay clumps with iron-stained coatings 1/8"± thick*
6.0- 8.5	*very fine to medium sand with a little silt; grayish brown*

A8

0- 0.9'	*sand; brown; road ditch fill*
0.9- 4.2	*medium to coarse sand with scattered fine gravel; brown*
4.2- 6.5	*fine to coarse sand; buff with a light gray layer of micaceous sand*
6.5- 7.0	*fine to medium sand; poorly sorted; buff to tan*
7.0- 8.0	*fine to medium sand; mottled reddish brown to dark brown and buff to tan*
8.0- 9.0	*very fine to medium sand; poorly sorted; buff to tan*

A9

0- 0.5'	*dark brown sandy topsoil*
0.5- 1.2	*fine to medium sand; rusty brown*
1.2- 7.5	*fine to medium sand; some finer sand and a little coarse sand mixed in layers*
7.5- 9.5	*sand as in 1.2'-7.5', except more poorly sorted; some very fine sand and a little gray silt*

A10

0- 0.8'	dark brown sandy loam topsoil
0.8- 1.5	medium to coarse sand; rusty brown
1.5- 7.5	fine to medium sand; buff to light gray
7.5- 8.0	very fine to fine and some medium sand with a clayey silt layer (probably less than 1" thick); mottled buff-gray and reddish brown
8.0- 9.0	fine to very fine sand; mica present but not abundant; buff to light gray

A11

0- 0.5'	light brown topsoil and roots
0.5- 1.5	fine to medium sand; rusty brown
1.5- 7.0	fine to medium sand becoming more poorly sorted with depth; buff to tan
7.0-10.3	very fine to coarse sand with scattered gravel; buff to tan

A12

0- 0.8'	brown sand loam topsoil
0.8- 7.0	fine to medium sand; relatively well sorted; buff to tan
7.0- 8.0	very fine to fine sand with some medium sand; buff to tan
8.0- 9.0	same as 7'-8', except mottled, i.e., reddish brown and buff to tan intermixed
9.0-10.7	same as 7'-8'

APPENDIX II--Auger Hole Logs (see Fig. 2 for location)
by Robert G. Gerber, Certified Geologist

Date of Drilling: 18 December 1980

A13--3' northeast of hole #A8, hit water between 6 and 11 feet, el. 67'

0- 1'	olive	fine	to	medium	sand		
1- 6'	"	"	"	"	"	, few gravel	
6-11'	"	"	"	"	"		
11-16'	"	"	"	"	"		
16-21'	"	"	"	"	"	, but slightly finer than 0-16'	
21-26'	"	"	"	"	"		
26-31'	brown	"	"	"	"		
31-36'	"	"	"	"	"		
36-41'	"	"	"	"	"		
41-46'	"	"	"	"	"		
46-51'	"	"	"	"	"		(no refusal, no sign of clay)

A14--35' east of Black Alder Brook, on south shoulder of Foreside Rd., el. 22'

0- 1'	gravelly sand and cobble road fill
1- 6'	dark brown silty fine sand, some medium sand, hit water
6-11'	" " " " " " "
11-16'	olive " " "
16-21'	" " " "
21-26'	" " " "
26-31'	" " " "
31-36'	" " " "
36-41'	gray clayey fine sand
41-46'	" " " "
46-51'	" silty clay (no refusal)

A15--60' west of Foster Brook in northeast corner of Sportsmen Club parking lot, el. 29', hit water between 6 and 11'

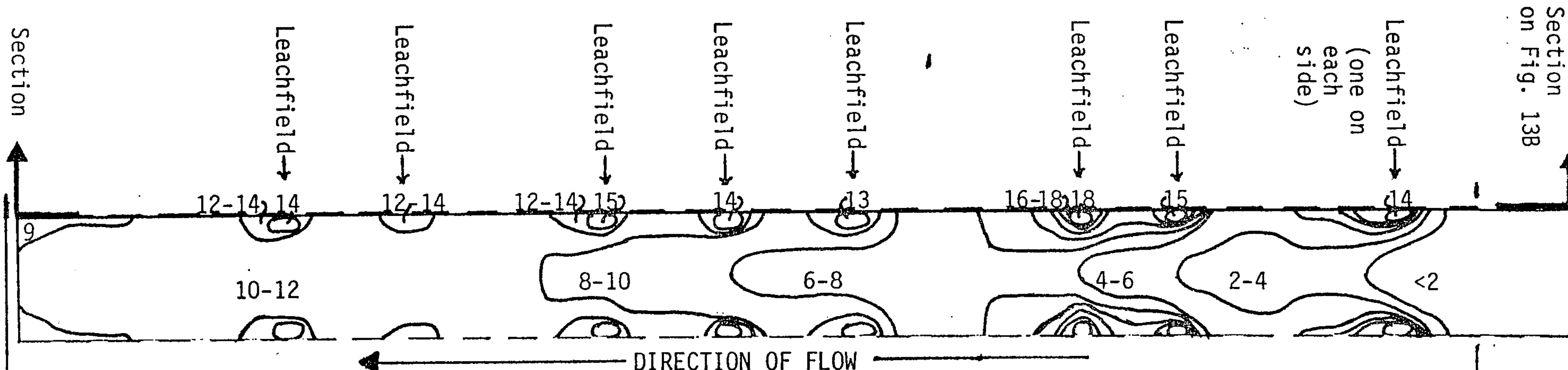
0- 1'	gravel parking lot fill
1- 6'	silty medium to fine sand
6-11'	silty fine sand
11-16'	fine sand
16-21'	" "
21-26'	" "
26-31'	very fine sand
31-36'	" " "
36-41'	" " "
41-46'	" " " , hit hard layer at 41'-42'
46-51'	clayey very fine sand (no refusal)

A16--75' west of A9, el. 62', hit water between 6' and 11'

0- 1'	loamy sand
1- 6'	fine to medium sand
6-11'	fine sand, little silt
11-16'	" " " "
16-21'	silty fine sand
21-26'	" " "
26-41'	" " "
41-46'	" " " , more fines than 16-46'
46-51'	" " " " " " " (no refusal)

A17--100' west of Foster Brook, on Stuart southern prop. line, el. 64'
hit water between 11' and 16'

0- 1'	*fine sand*
1- 6'	*" "*
6-16'	*" "*
16-21'	*silty fine sand*
21-26'	*" " "*
26-31'	*clayey fine sand*
31-46'	*soft gray silty clay (no refusal)*

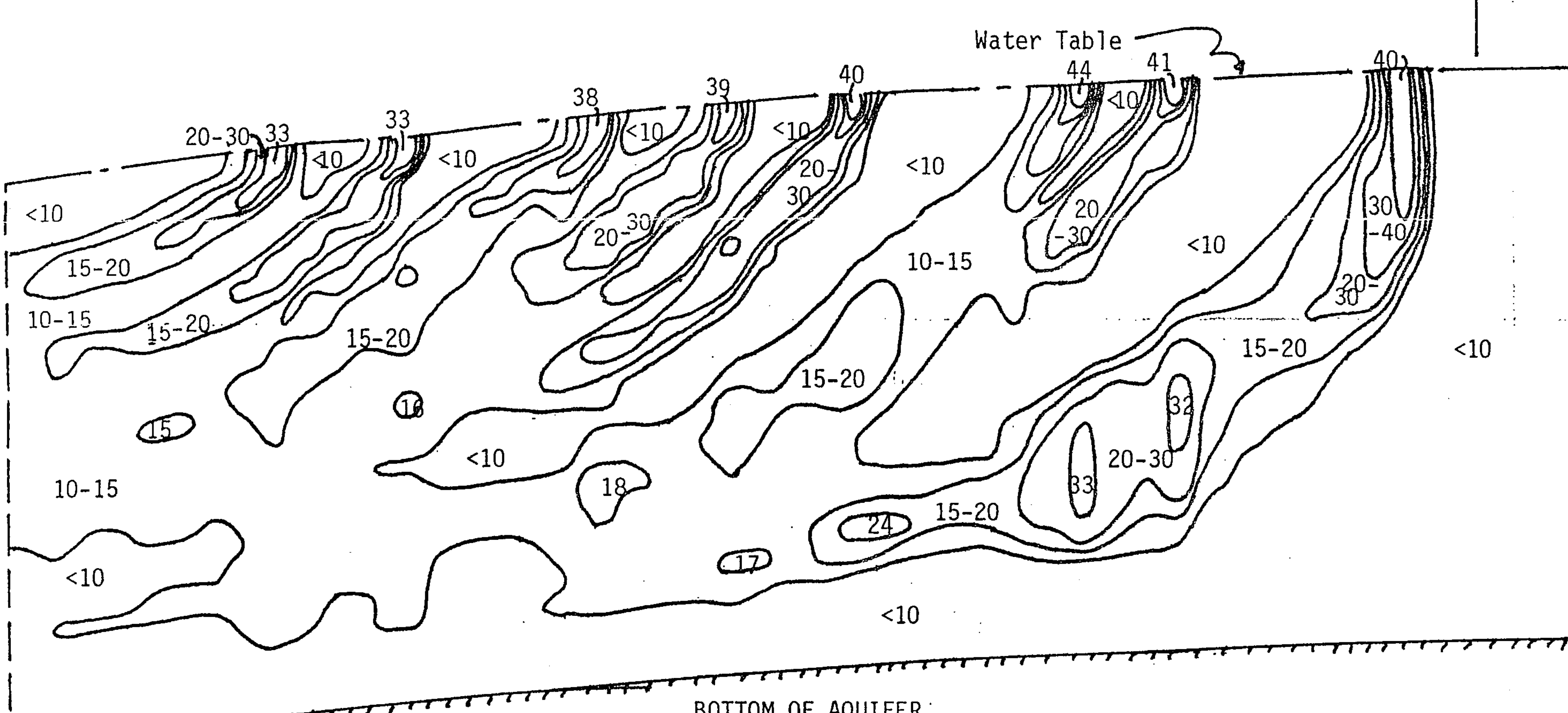


Run 51	Run 50	Run 49
Simulation Period = 8 years	Simulation Period =10 years	Simulation Period=10 years
Average gradient = 0.0116	Average gradient = 0.0072	Average gradient = 0.0036
Mass Balance Error = -0.04%	Mass Balance Error = -0.05%	Mass Balance Error = -0.2%
Chem. Mass Bal. Error = 1.8%	Chem. Mass Bal. Error = 3.4%	Chem. Mass Bal. Error=11.2%

Fig. 13A--Plan View of Vertically-averaged pollutant concentrations as a percent of leachfield Contaminant Concentration

Scale: 1"=200' Grid Element Dimensions: 40' in direction of flow; 20' normal to flow
Assumed leachfield size: 20'x40' with long dimension oriented parallel to flow
Aquifer thickness: 72'
Precipitation recharge rate: 6×10^{-8} ft/sec
Transmissivity: 0.015 square feet per second
Longitudinal Dispersivity: 30'
Transverse Dispersivity: 9'

Gradients developed for the case of no line sinks assumed in development (see Run 39)



Run 45	Run 44	Run 43
Simulation period = 8 years	Simulation period = 10 years	Simulation period=10 years
Average gradient = 0.0116	Average gradient = 0.0072	Average gradient = 0.0036
Mass Balance Error = -0.1%	Mass Balance Error = -0.1%	Mass Balance Error= -0.2%
Chem. Mass Bal. Error = 13.1%	Chem. Mass Bal. Error = 26.1%	Chem. Mass Bal. Error=32.1%

Fig. 13B--Vertical Cross-section view in direction of flow along centerline of 8 aligned leachfields; Section taken as shown from Fig. 13A and concentrations given in percent of leachfield contaminant concentration

Horizontal scale: 1"=200' Vertical scale: 1"=20'
Grid Element Dimensions: 40' horizontally, 4' vertically
Aquifer thickness 72'
Longitudinal and Transverse Dispersivity: 0' (no dispersion included in vertical plane)
Horizontal permeability = 2.1×10^{-4} ft/sec; Vertical permeability = 2.1×10^{-5} ft/sec

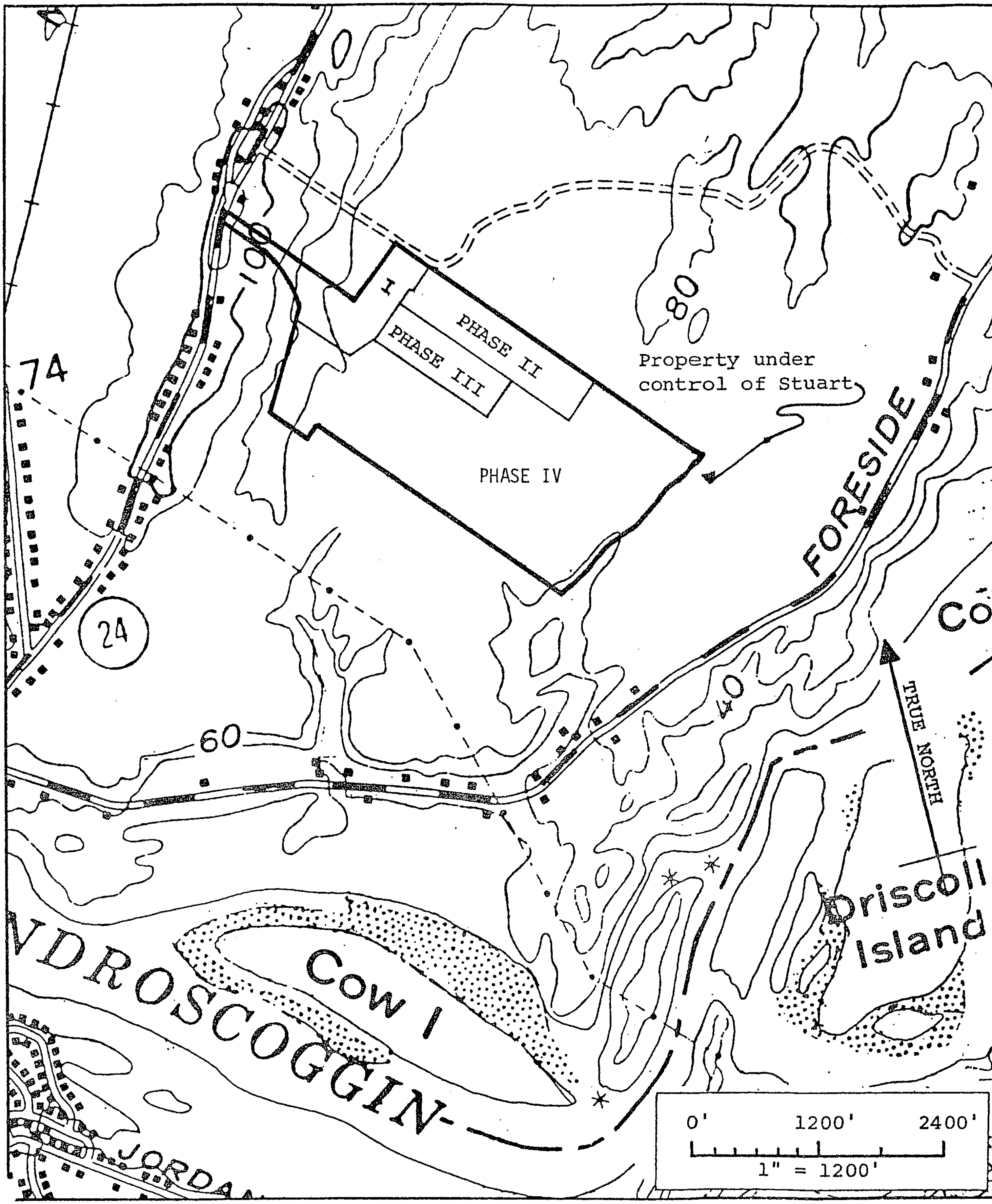


Fig. 1--Location Map: Bay Park, Topsham, Maine

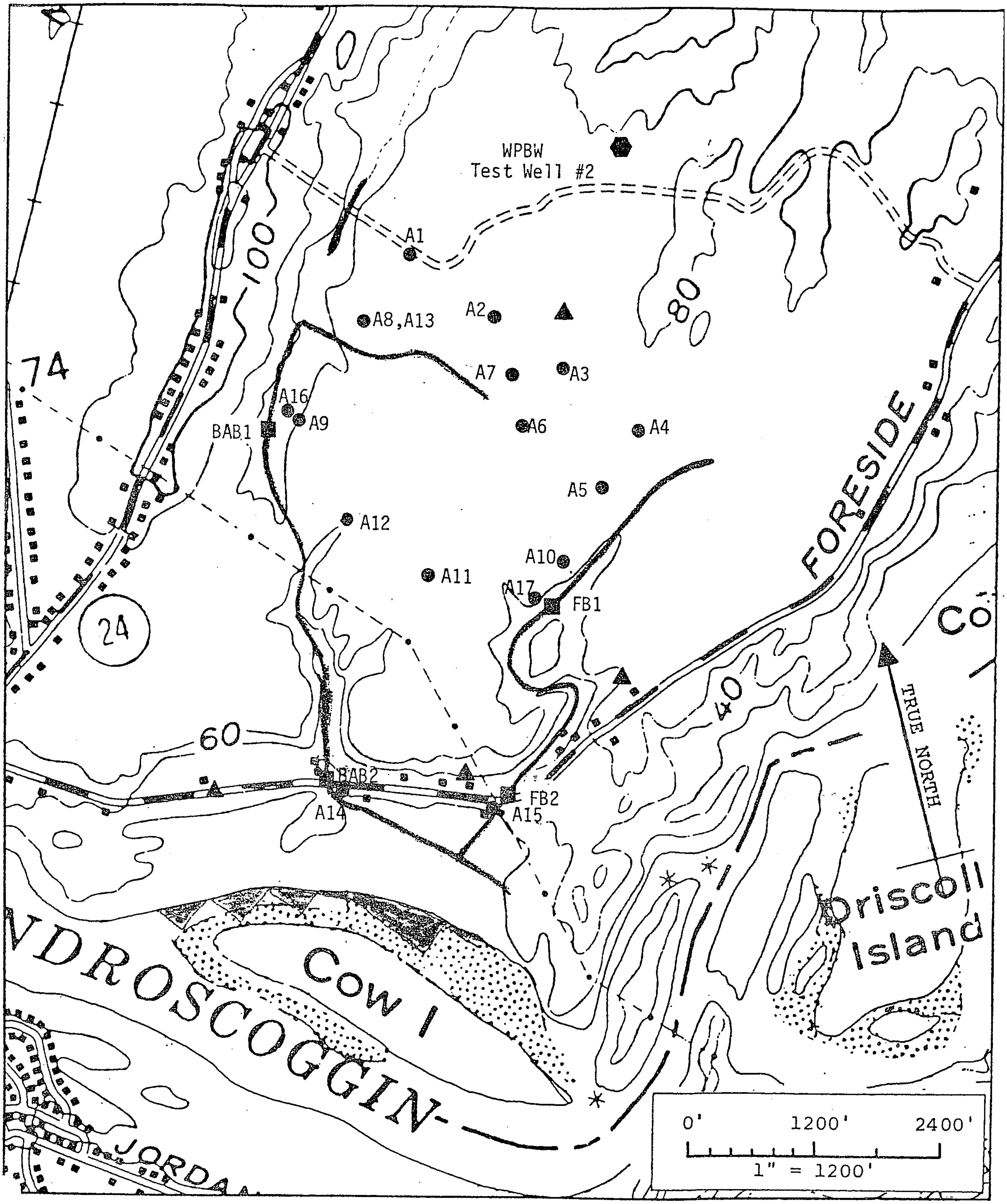


Fig. 2 --Location of Geologic Data Points; Run #36 Line Sinks; Water Quality Test Points

Line sinks shown in red:

● A14 --location recent auger boring

▲ data from Prescott, 1967

BAB1 --stream water quality test point

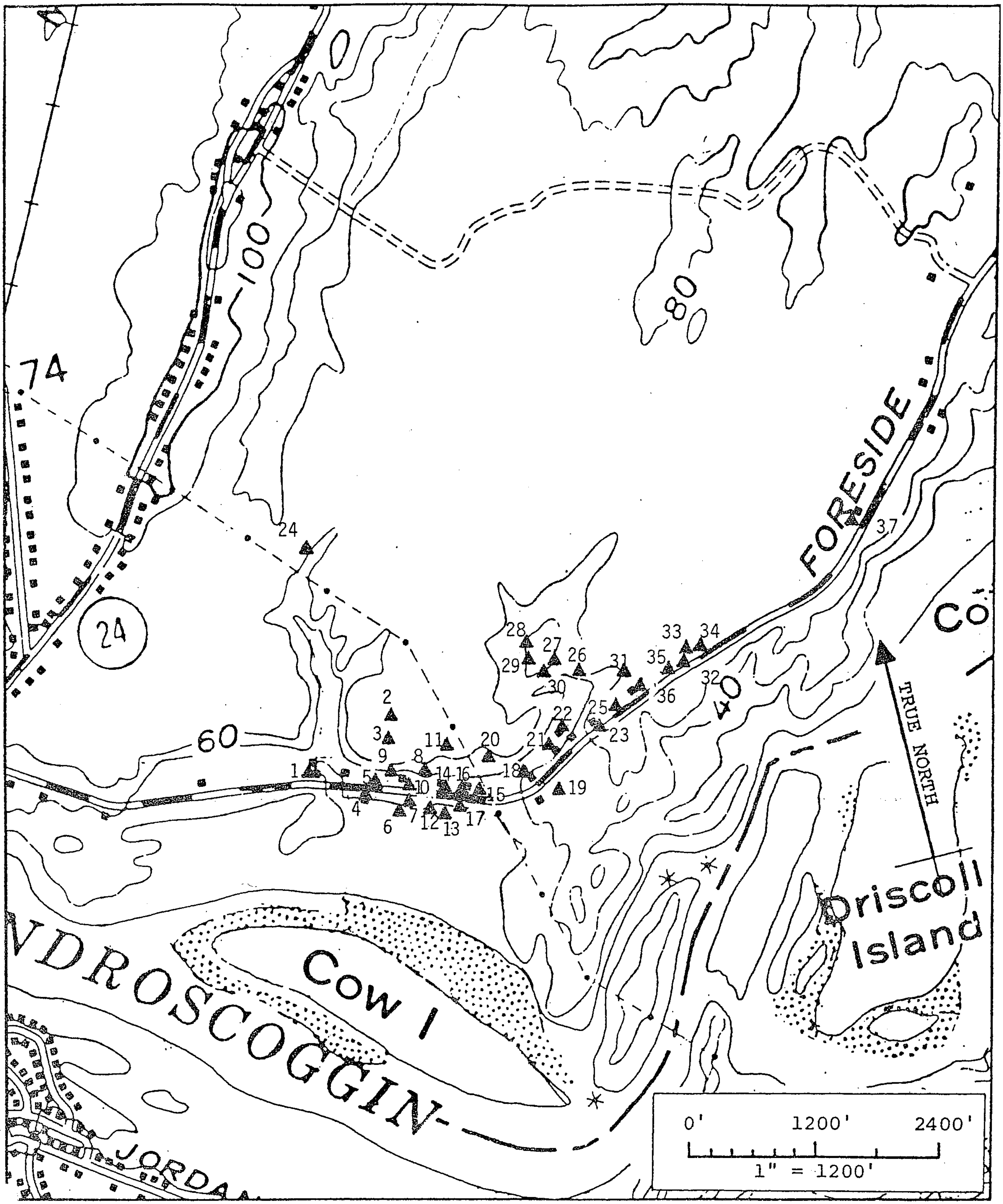


Fig. 3--Location of Private water Wells surveyed for this Study

See Table 3 for description of each well

▲ 19 Location of well; number is keyed to data in Table 3

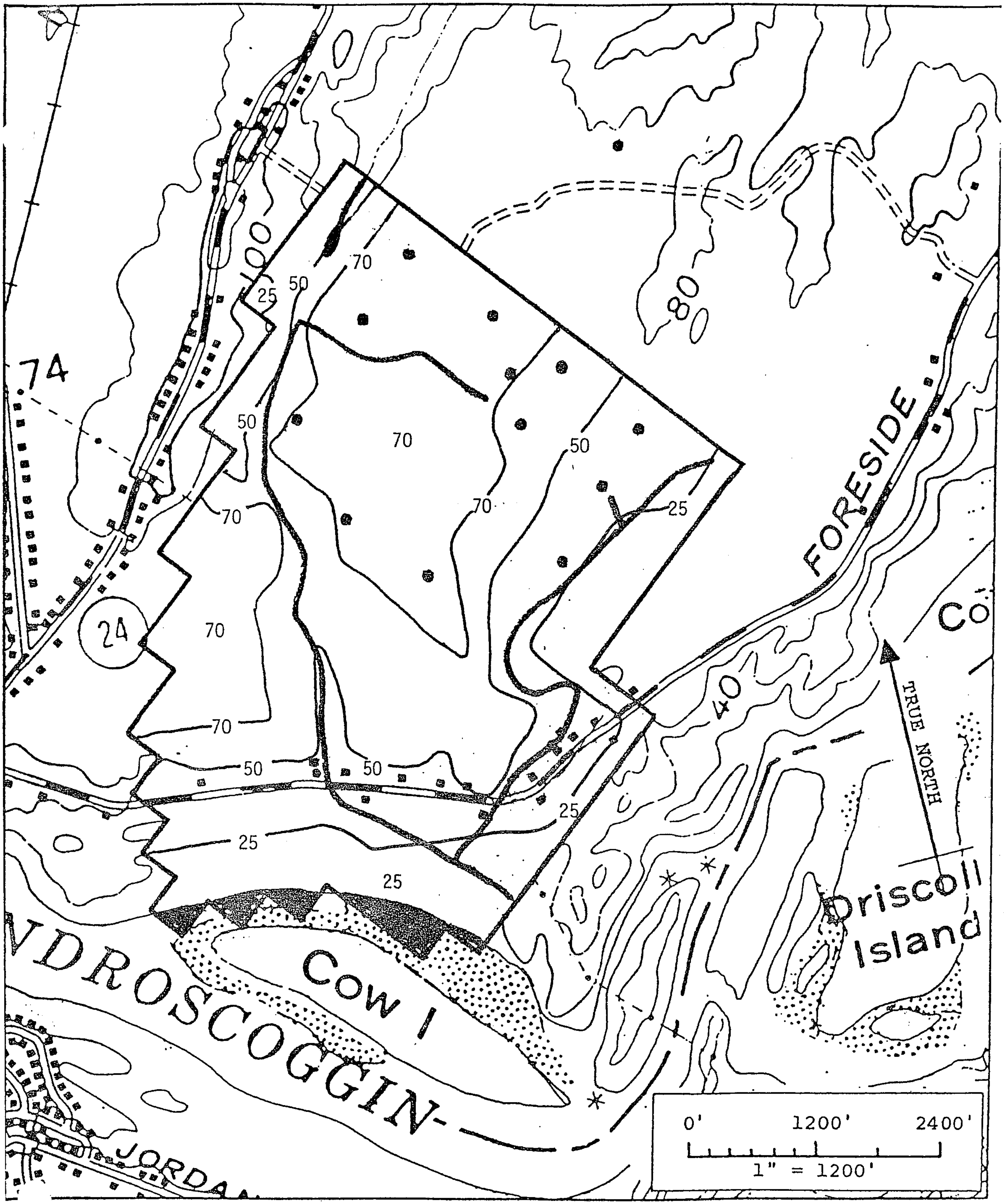


Fig. 4--Aquifer Thickness

Contour values given in feet of saturated thickness under average recharge conditions

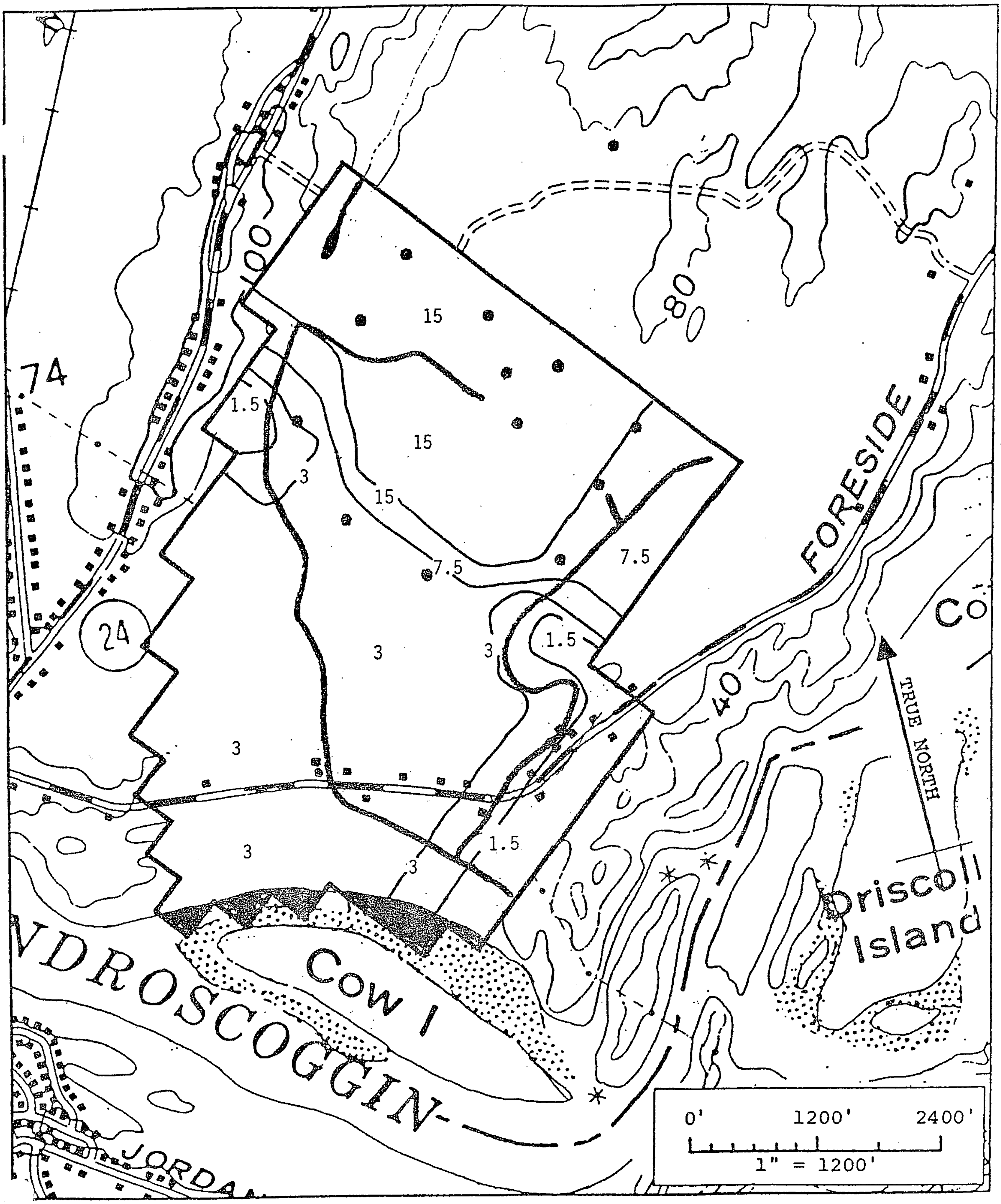


Fig. 5--Aquifer Transmissivities

Multiply contour value by 10^{-3} to obtain transmissivity in square feet per second

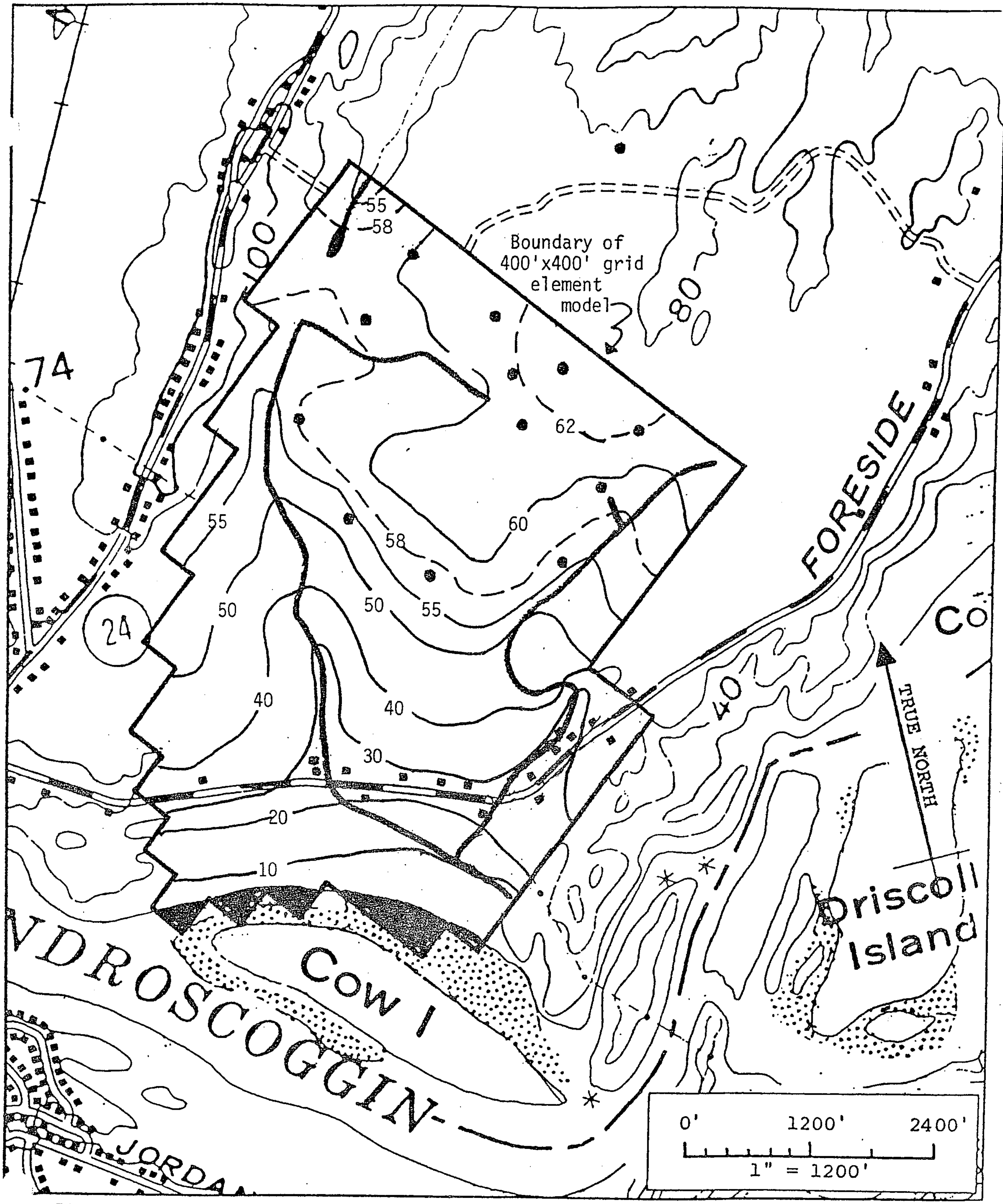


Fig. 6--Ground Water Contours, Steady State Condition after Phases I-IV Developed

Precipitation recharge: developed areas= 6×10^{-8} ft/sec; undeveloped = 7×10^{-8} ft/sec
Mass Balance Error: 0.0001% Assumes Line Sinks in development as shown (Run 40)
Ground water contours given in feet above Mean Sea Level

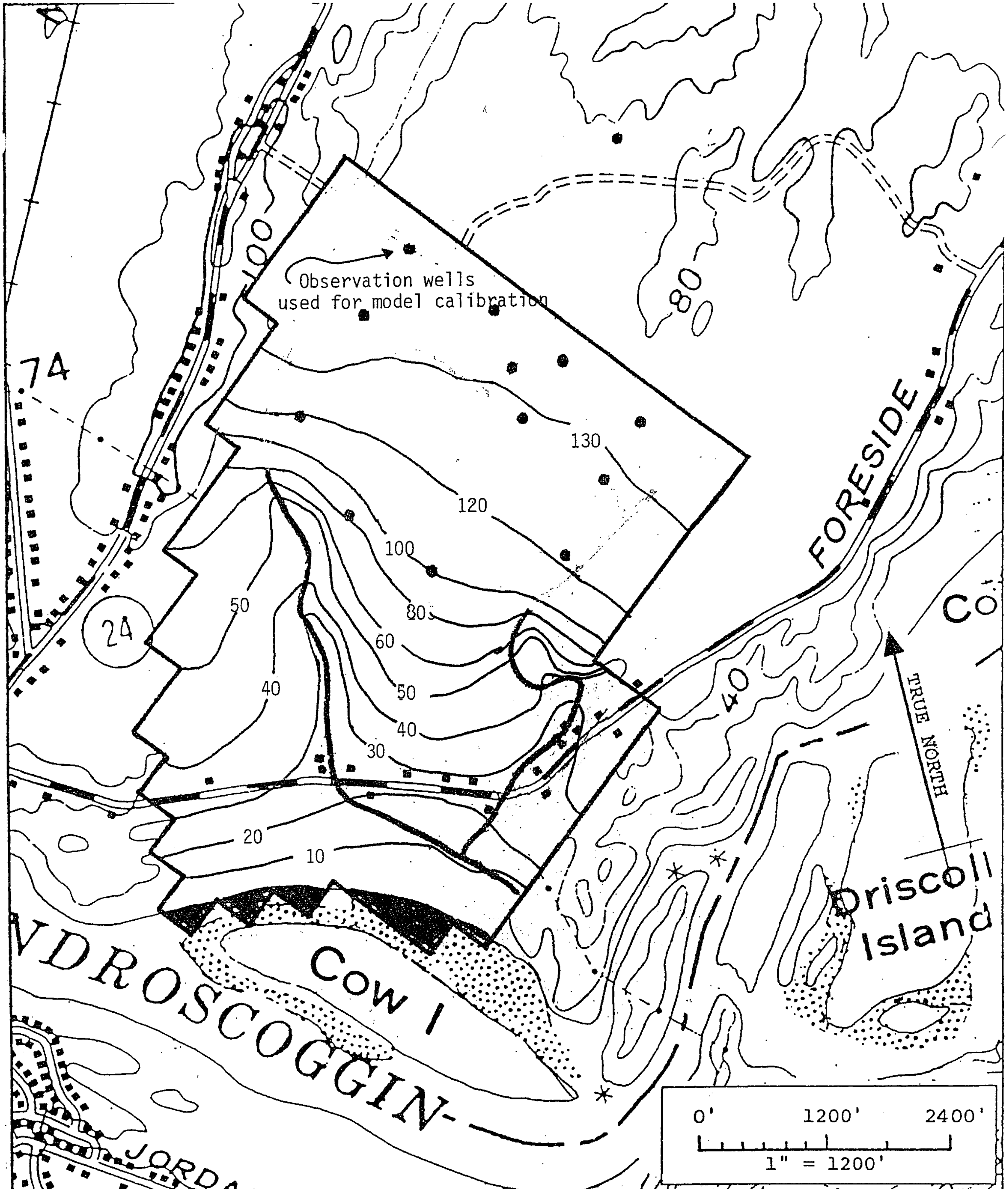


Fig. 7--Ground Water Contours, Steady State Condition after Phases I-IV Developed

Precipitation recharge: developed areas= 6×10^{-8} ft/sec; undeveloped = 7×10^{-8} ft/sec
Mass Balance Error: 0.0002% Assumes no line sinks in development (Run 39)
Ground Water Contours given in feet above Mean Sea Level

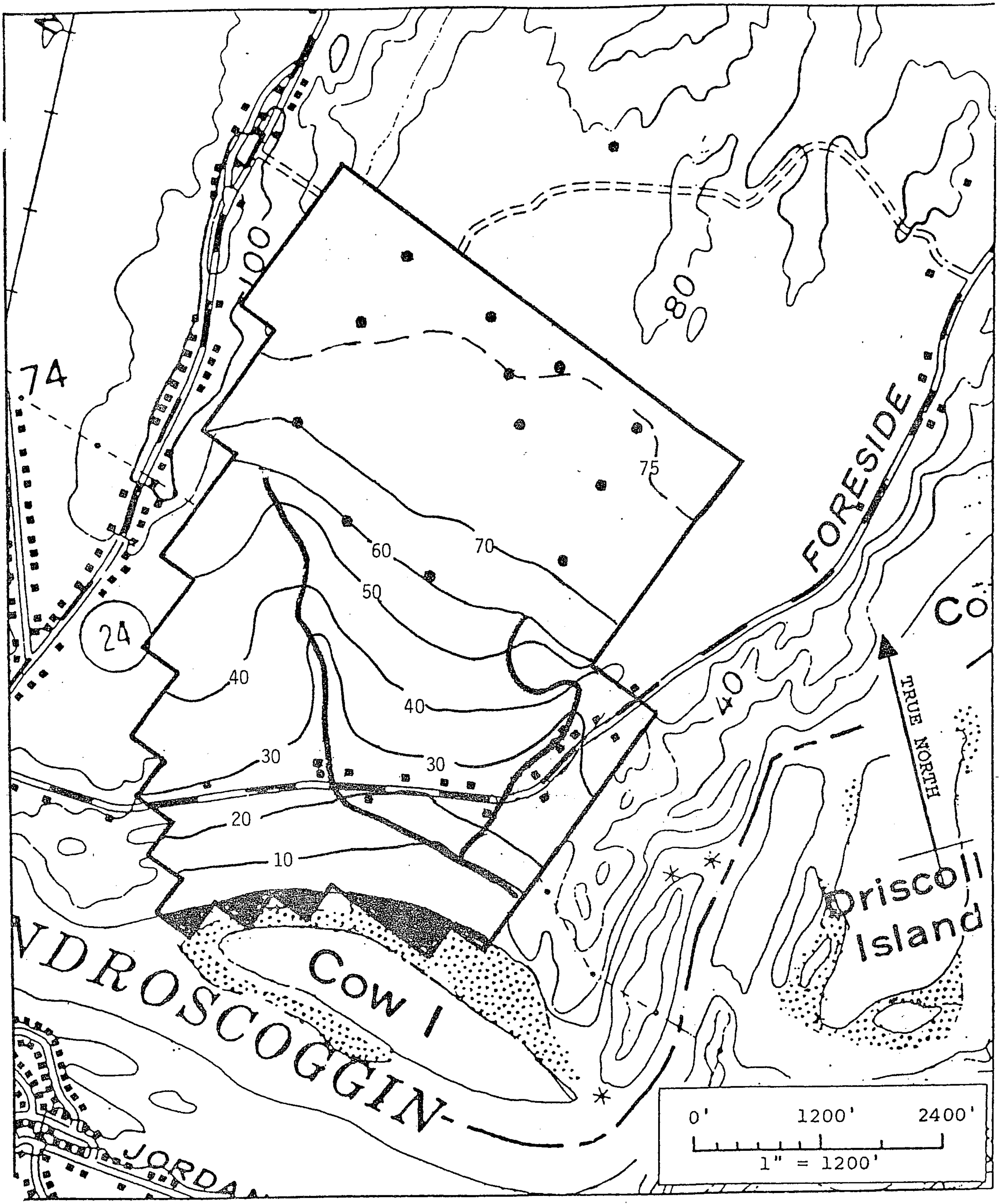


Fig. 8--Ground Water Contours after 2-year drought with Line Sinks taken out

Precipitation Recharge = 4×10^{-8} ft/sec   Initial ground water contours as shown on Fig. 6
Mass Balance Error: 0.0007% Assumes line sinks as shown (Run 41)--Values in feet (MSL)

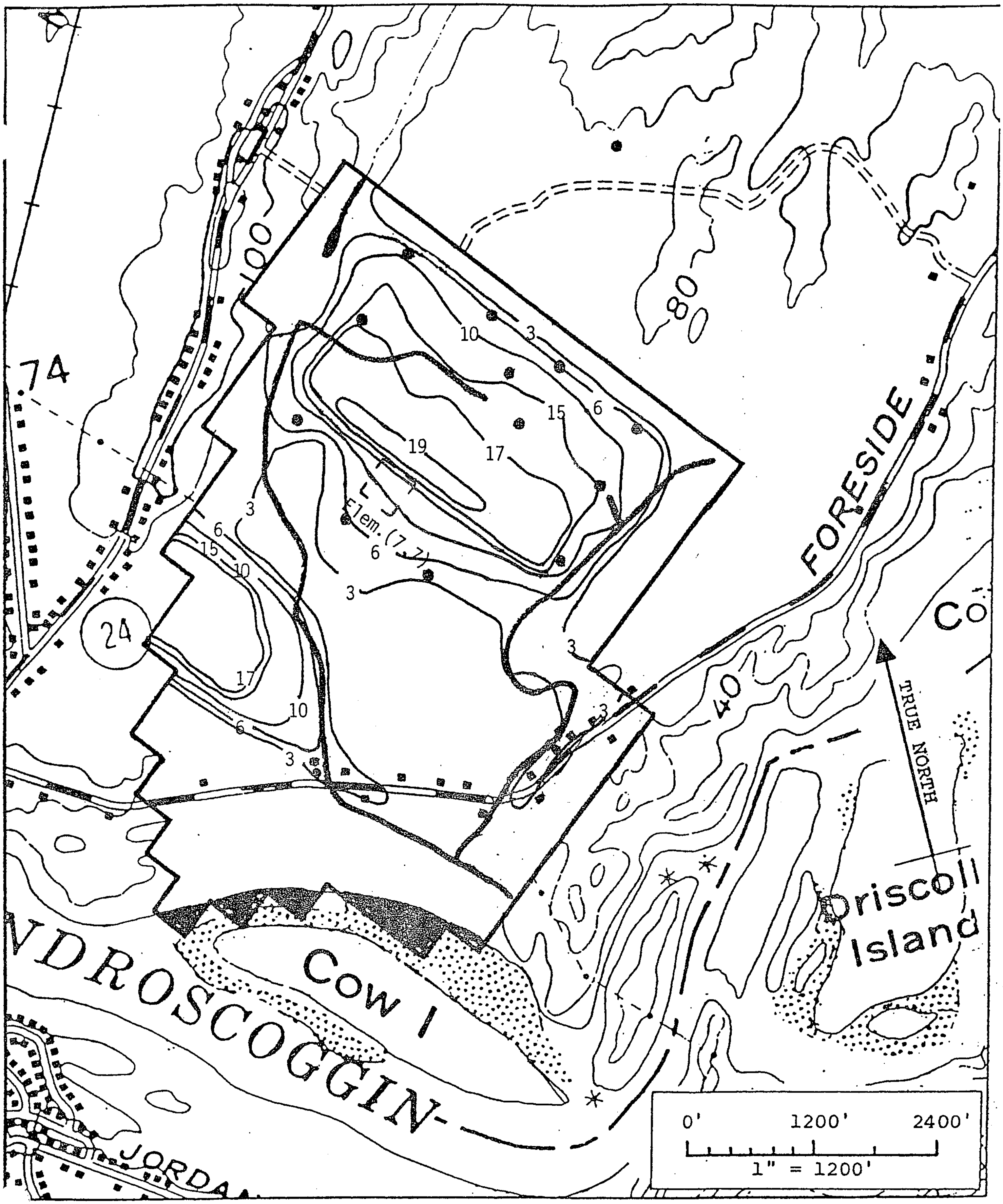


Fig. 9--Isocon lines after 40 years; Percent of Leachfield contaminant concentration

Same conditions as Fig. 6; line sinks assumed in development as shown; adjacent development taken into account; Chemical Mass Balance Error: 2.6% (Run 40)
Grid element (7,7) straddles Stuart's southern boundary; see Fig. 12B

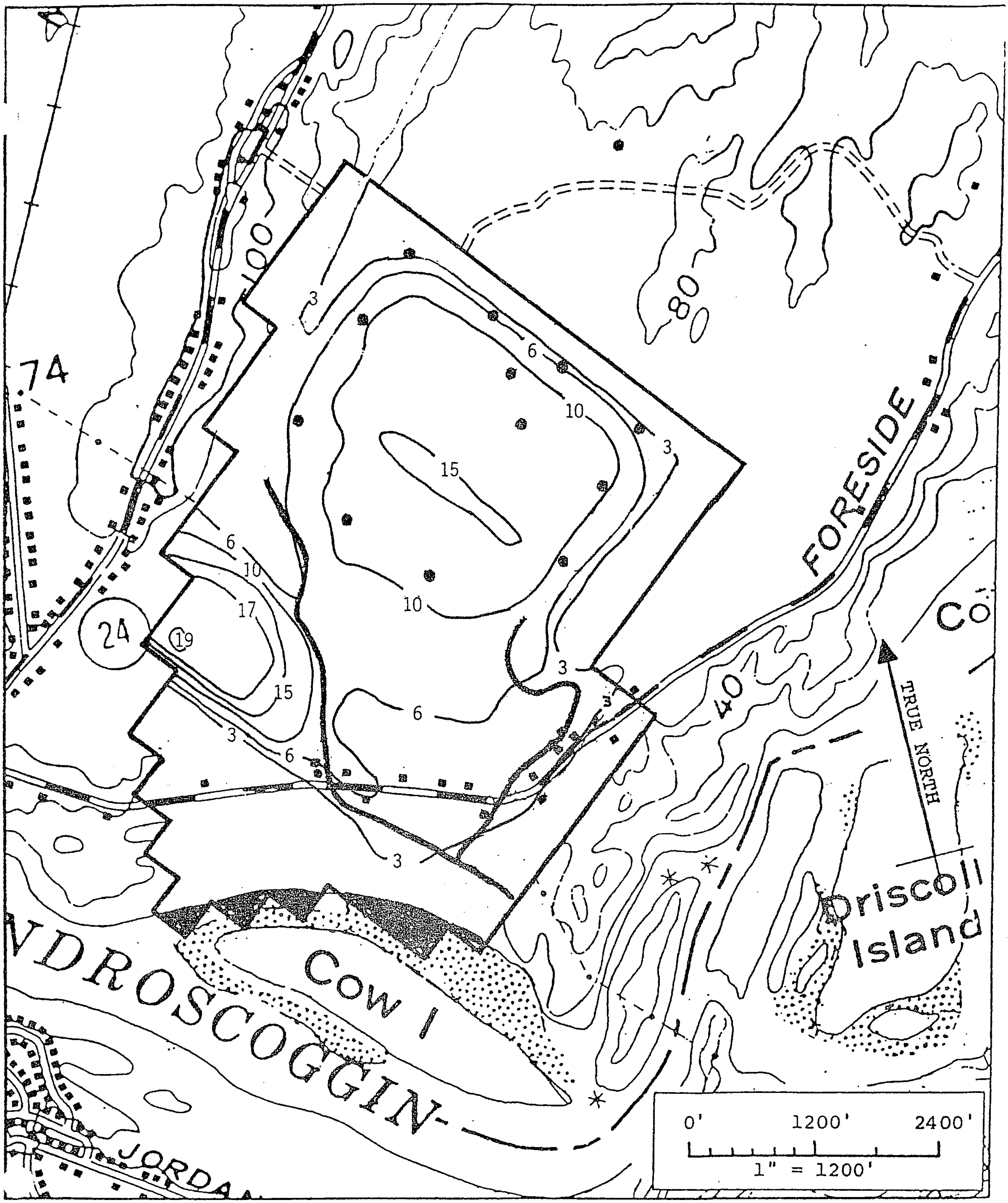


Fig. 11--Isocon Lines after 2-year drought starting with water table as shown in Fig. 6 and Aquifer contaminant distribution as shown on Fig. 10

Chemical Mass Balance Error: -11.8% (Run 41); values given in percent as in Figs 9 & 10

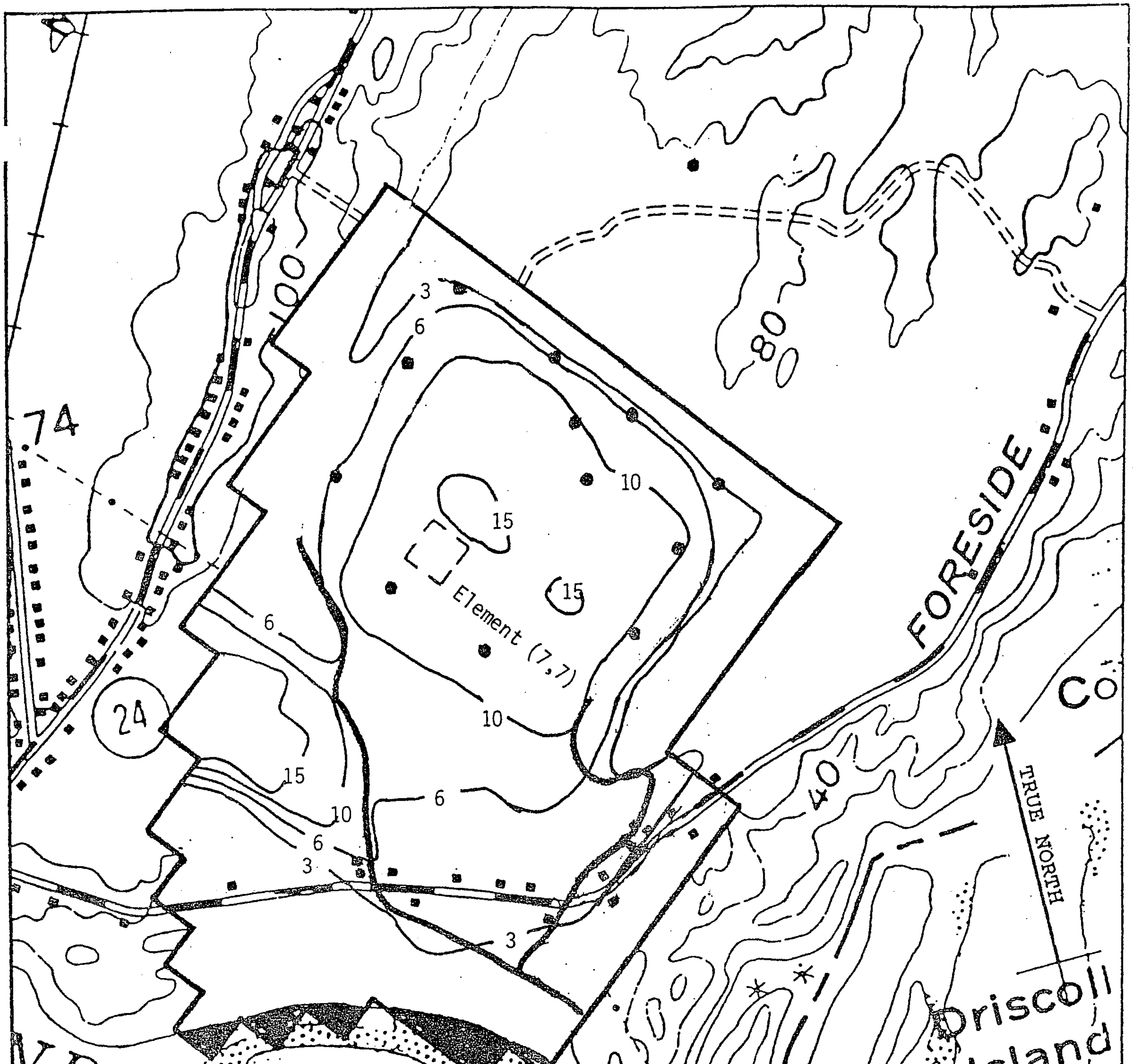


Fig. 10--Isocon Lines after 30 Years; Percent of leachfield contaminant concentration

Same conditions as Fig. 7; no line sinks assumed in development; adjacent development taken into account; Chemical Mass Balance Error: 1.6% (Run 39)
Grid Element (7,7) straddles Stuart's southern boundary: see Fig. 12A

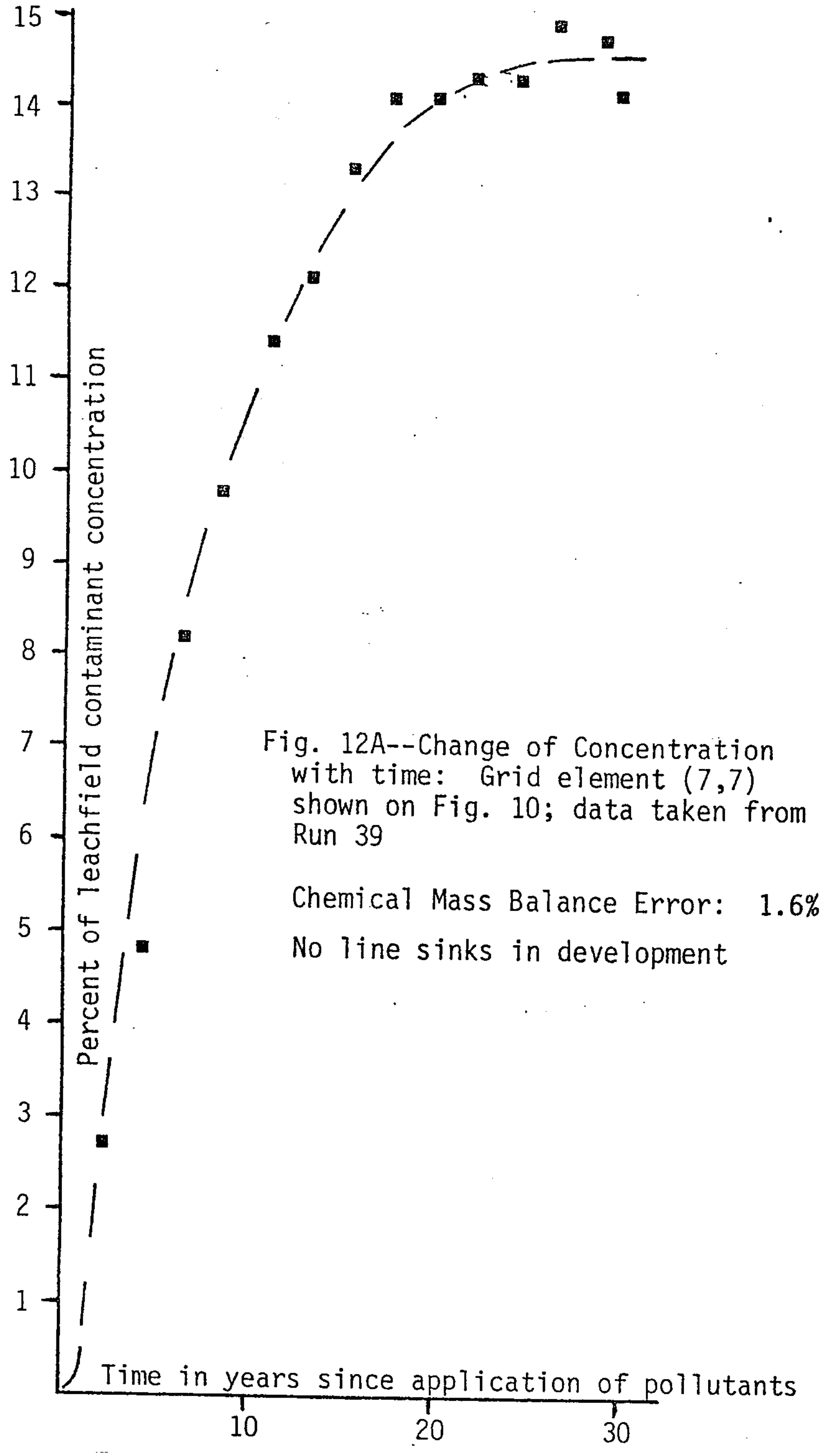


Fig. 12A--Change of Concentration with time: Grid element (7,7) shown on Fig. 10; data taken from Run 39

Chemical Mass Balance Error: 1.6%

No line sinks in development

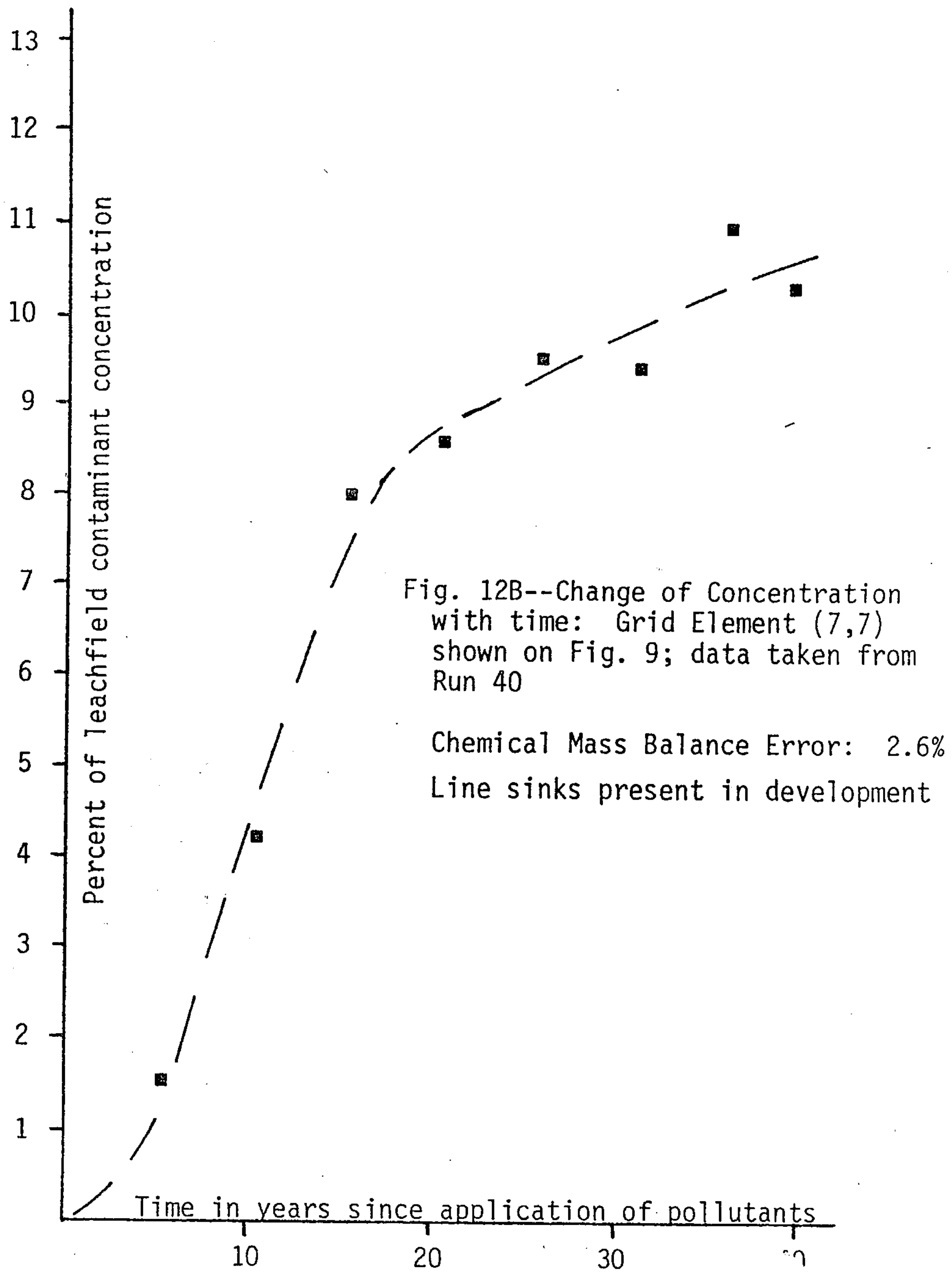


Fig. 12B--Change of Concentration with time: Grid Element (7,7) shown on Fig. 9; data taken from Run 40

Chemical Mass Balance Error: 2.6%

Line sinks present in development