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# Groundwater Travel Times near Spreading Ponds: Inferences from Geochemical and Physical Approaches

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**Abstract:** Groundwater travel time is an important criterion for regulating managed aquifer recharge (MAR) operations because of its relationship to water quality. Here, three complementary methods for determining travel times are examined. Sulfur hexafluoride (SF<sub>6</sub>), a gas tracer, was injected into 23 spreading basins at the Montebello Forebay MAR operation (Los Angeles County, United States) and monitored at ten monitoring and 18 production wells within 150 m. Over 2 years, SF<sub>6</sub> was detected at nine monitoring and 11 production wells. Travel times showed a significant relationship with depth, but not with horizontal distance or pumping rate. A pumping influence was apparent as the tracer arrived sooner at production wells than at monitoring wells of similar depth. In the unconfined aquifer, estimated hydrogeologic travel times were <0.2 years (<10 weeks) and agree with the SF<sub>6</sub> data. However, in the confined aquifers, estimated travel times were >4 years and the agreement with the SF<sub>6</sub> travel times was poor. At the seven production wells with SF<sub>6</sub> detections, leakage through low permeability layers leading to earlier tracer arrival provides a likely explanation. All tritium/<sup>3</sup>He ages at production wells are greater than 10 years; this data combined with the SF<sub>6</sub> results indicate the wells produce a mixture of young and old groundwater.

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**CE Database subject headings:** Groundwater management; Ponds; Travel time; Aquifers; Water quality; Artificial recharge; Tracers.

## Introduction

Groundwater has been a primary source of potable and irrigation water for centuries. In the western United States and other semi-arid regions where water is limited, the efficient management of this vital resource is critical. The projected growth in population combined with uncertainties associated with a changing climate is producing unprecedented stresses on water supplies already stretched to their limits. In addition to conservation, new solutions are needed to meet future water demands in these areas.

A recent advancement in groundwater/surface water management aimed at augmenting water supplies is managed aquifer recharge (MAR). This practice, which is also known as groundwater banking, aquifer replenishment, artificial recharge, bank filtration, and aquifer storage and recovery (ASR), consists of recharging imported water, reclaimed (recycled) wastewater, or surplus runoff into aquifers for storage and later extraction. A number of different designs, including spreading basins (infiltration ponds), engineered natural stream channels, and injection wells are used to facilitate the recharge process (Bouwer 2002).

The water is often recharged into aquifers depleted by overproduction and the hydraulic effects of MAR (e.g., water table rise) are usually quick and considered beneficial. As a result, MAR has become an important method for the combined management of groundwater and surface water in many areas of the world with new operations continually implemented (Haarhoff and Van de Merwe 1996; de Jonge et al. 2002; Mills 2002; Tufenkji et al. 2002; Massmann et al. 2004; Dillion 2005; NRC 2007).

Fundamental issues concerning MAR are as follows: (1) the hydrogeology and engineering considerations of site evaluation, recharge method, and clogging (see Bouwer 2002); (2) source water supply; (3) water quality; and (4) the potential impairment of the aquifer and native groundwater supply. The latter three issues are often linked and center on concerns about supplying safe drinking water. The introduction of disinfection byproducts, infective microorganisms, and organic compounds with unknown health risks into groundwater supplies is a significant concern (NRC 2007). This is especially true at MAR sites, where either reclaimed wastewater or urban runoff is a principal source of water for the operation.

It is likely that in the future, reclaimed water will become a larger portion of the source water supply at most MAR operations because the availability of imported and local river water may shrink due to shifts in climate and the diversion of this water for new uses such as maintaining riparian ecosystems. Because water quality concerns are raised when reclaimed wastewater is a large portion of the source water, it is paramount to understand the fate and transport of potential contaminants near MAR sites. Only from this understanding can robust and appropriate regulations be developed. Results of detailed water quality studies near MAR operations have shown that the most important hydrologic parameters are travel time and distance.

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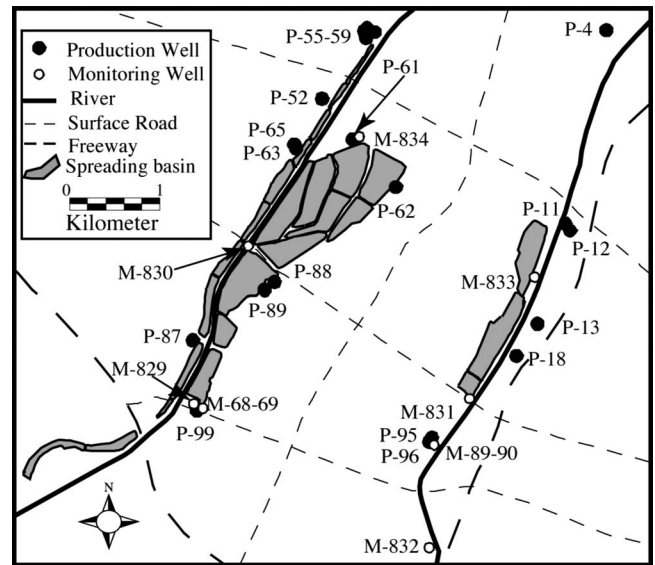
A large number of potential contaminants, such as many organic compounds and most infective microorganisms, are naturally removed or become inactive with time and distance in the subsurface (Yates and Yates 1987; Fox et al. 2001; Drewes et al. 2002; Hiscock and Grischek 2002). These processes, which are collectively referred to as soil aquifer treatment (SAT) or geopurification, are considered an additional benefit of MAR, especially when reclaimed water is a significant portion of the source water (Mansell and Drewes 2004; Dillion 2005). As a result, regulations of MAR operations often specify a minimum travel time and distance before recharge water can be produced for drinking water. For instance, in 2006, the California Department of Public Health (CDPH), which regulates operations near MAR sites that recharge reclaimed wastewater, updated their water reuse draft rules to specify that groundwater supply wells must be more than 150 m (500 ft) from MAR facilities (CDPH 2007). The rules do allow for a reduction of the horizontal distance requirement if the water agency can document that the travel time is greater than 6 months (CDPH 2007).

This study was designed to examine travel time to production wells within 150 m of spreading basins, the critical distance defined by the CDPH reuse rules. The study, which was conducted at the Montebello Forebay Spreading Grounds in Los Angeles County, Calif., compared travel time estimates using three independent but complementary methods: a deliberate tracer experiment, hydrogeologic calculations, and tritium/helium-3 ( $T/{}^3\text{He}$ ) dating. In particular, this study tests the validity of the travel distance requirement of the CDPH reuse rules considering that for many nearby production wells, depth, not horizontal distance, determines the length of groundwater flow paths. It also shows how the different methods elucidate different aspects of the flow system, all of which are needed for developing better regulations.

### Study Location

The Montebello Forebay Spreading Grounds are located in Los Angeles County, Calif., within the principal recharge area for the Central Groundwater Basin. The groundwater basin is bounded by the Hollywood Basin and the Elysian, Pepetto, Merced, and Puente Hills in the north, the Orange/Los Angeles county line to the east, and the Newport-Ingelwood fault and structural uplift in the south and west. This alluvial basin is filled with Holocene and Pleistocene unconsolidated stream, flood, marine, and alluvial fan deposits overlying Pliocene marine terrace and alluvial sediments. Water-producing sediments are Pleistocene to recent in age and have little deformation (CDWR 1961). The hydrostratigraphy consists of layers with a high percentage of sand and gravel interbedded with aquitards of clay to fine-grained silt. The spatial extent of any given layer is poorly known (Bookman-Edmonston Engineering 1994). The spreading grounds overlie numerous aquifers; near-surface unconfined aquifers merge with deeper aquifers in this area making this location ideal for groundwater recharge for the Central Basin. High production of groundwater is possible due to the permeable sediments, the relatively undeformed state of the sediments, and a low topographic profile.

In the early 1900s, more than half the production wells in the Central Basin were artesian. As the population increased, severe overdrafting caused groundwater levels to drop significantly, eventually lowering them to below sea level. This resulted in salt-water intrusion along portions of the coast. Water table levels reached their lowest level in 1957. Although limits on production were imposed in the early 1960s, extraction continued to exceed natural recharge. Artificial recharge of groundwater using im-



**Fig. 1.** Map of Montebello Forebay spreading basins and wells sampled during study. Rio Hondo basins and river lie to west of San Gabriel basins and river.

ported water began in 1954 and using reclaimed water in 1962. While groundwater levels have risen and are relatively stable today, they have not returned to previous levels (WRD 2005). The current groundwater levels in the Central Basin range from 50 m above sea level (asl) to 37 m below sea level (bsl).

The MAR site contains 23 shallow (<4 m deep) spreading basins adjacent to the Rio Hondo and San Gabriel rivers with a total wetted area of about 200 ha (Fig. 1). Due to urbanization, recharge in the area is limited to these basins with some additional recharge occurring through unlined portions of the San Gabriel River; the Rio Hondo is lined with concrete throughout the study area and does not recharge the aquifer. During our investigation, travel times to ten monitoring and 18 production wells were examined. With the exception of only two, the wells are within 150 m of the spreading basins and fail to meet the horizontal distance criterion of current CDPH draft regulations. Depth of production and screen length vary considerably between wells. Screen lengths range from 6 to 23 m for monitoring wells and from 21 to 159 m for production wells. Tops of well screens are as shallow as 8 m and bottoms are as deep as 210 m (Table 1).

During the last decade, approximately  $1.6 \times 10^8 \text{ m}^3/\text{year}$  of surface water was recharged annually at the Montebello Forebay Spreading Grounds. About 40% of this water was natural storm runoff, 25% was imported water, and 35% was reclaimed water. In general, groundwater flows southwest from the Montebello Forebay in the Central Basin (Bookman-Edmonston 1994). Water table elevation fluctuated below the Montebello Forebay during the study in response to the seasonal recharge and use cycle. More water is recharged during the winter months when season precipitation produces significant runoff.

## Principles of Travel Time Determination

### Deliberate Tracer Experiment

In deliberate tracer experiments at MAR operations, tracer is added to surface or recharge water and its arrival time is deter-

**Table 1.** Summary Well Information and Travel Time Data

WRD ID	Ground elevation (m)	Depth <sup>a</sup> (m)	Distance from pond (m)	Screen length (m)	Mean production (m <sup>3</sup> /s)	Hydrogeology travel times (years)	T/ <sup>3</sup> He apparent ages (years)	SF <sub>6</sub> first arrival (years)
P-11	54	85	52	48	2.4	5.0	—	1.38
P-12	54	45	95	107	34.7	4.0	—	0.81
P-13	50	32	120	156	2.4	4.7	17	>2
P-18	49	52	77	159	6.5	5.6	41	1.00
P-52	61	63	98	37	26.5	1.9	19	>2
P-55	65	9	83	43	22.9	0.1	—	0.15
P-56	65	34	123	23	24.7	2.6	—	>2
P-57	65	34	123	22	23.5	2.6	20	>2
P-58	60	27	105	23	25.9	0.2	17	0.15
P-59	65	33	59	25	23.5	2.6	10	>2
P-61	54	41	18	77	10.6	0.1	6	0.15
P-62	53	86	66	50	17.6	11.7	—	1.00
P-65	55	14	77	90	23.5	0.1	10	0.31
P-87	51	61	40	122	17.1	4.4	—	>2
P-88	50	84	22	86	54.1	14.1	11	1.60
P-89	50	82	26	90	35.9	14.1	31	1.60
P-96	46	75	56	65	11.8	6.4	—	>2
P-99	45	68	18	112	5.3	5.7	7	0.81
M-68	44	83	19	6	—	—	8	1.77
M-69	44	40	19	6	—	—	2	0.54
M-89	46	73	31	6	—	—	3	1.60
M-90	46	31	31	6	—	—	7	0.54
M-829	48	14	4	19	—	—	—	0.08
M-830	50	12	43	12	—	—	—	0.23
M-831	49	17	3	6	—	—	—	0.15
M-832	44	13	37	23	—	—	—	>2
M-833	50	14	34	9	—	—	—	0.03

<sup>a</sup>Depth is determined from nearest pond bottom to top of the well screen.

mined directly through periodic sampling of wells. At each sampling location, initial and mean arrival times of the tracer can be determined by evaluating breakthrough curves, plots of concentration versus time. In homogenous aquifers, the initial and mean arrival times of tracer at narrow-screened monitoring wells represent the fastest and mean flow paths in the aquifer, respectively. In aquifers with preferential flow, tracer breakthrough curves are more complicated, often showing multiple peaks (Clark et al. 2004). Tailing is also evident on breakthrough curves and can represent the tracer reaching the well by slower flow paths or be indicative of retardation, which in the case of gas tracers can be due to trapped air (Fry et al. 1995; Donaldson et al. 1997; Vulava et al. 2002).

In the present study, SF<sub>6</sub> was chosen because it is an inexpensive, nonreactive gas and has been commonly used in deliberate tracer experiments at MAR sites (Gamlin et al. 2001; Clark et al. 2004, 2005; Avisar and Clark 2005). Because of its low toxicity (Lester and Greenberg 1950), it has been approved for use by the CDPH in potable aquifers. Laboratory and field experiments have shown that SF<sub>6</sub> transport is not retarded within saturated porous media (Wilson and Mackay 1996; Gamlin et al. 2001; Vulava et al. 2002). While retardation commonly occurs when trapped gas is present in the saturated zone (Fry et al. 1995; Donaldson et al. 1997; Vulava et al. 2002; Heilweil et al. 2004), Clark et al. (2004, 2005) found no significant retardation during field experiments below spreading ponds using multiple gas tracers. These observa-

tions were duplicated at the Rio Hondo spreading grounds by Quast et al. (2006), who showed that arrival times of xenon (as <sup>124</sup>Xe) and dissolved boron (as <sup>10</sup>B enriched borate) agree well; the gas tracer was not retarded relative to the ion tracer.

SF<sub>6</sub> has properties similar to fluorescent dyes and ionized substances but is less expensive, making it cost effective for tagging large bodies of water (>10<sup>6</sup> m<sup>3</sup>) without interference from density-induced flow (Istok and Humphrey 1995). In recharge facilities that rely on infiltration from spreading basins or rivers, loss of SF<sub>6</sub> at the air-water interface from gas exchange can be problematic. In order to define the input function of tracer to the groundwater at these settings, careful monitoring of surface water is required. This loss, however, does not interfere with the tracer results, as the primary purpose of the experiment is to determine the tracer arrival time, not tracer concentration at the wells.

### Hydrogeologic Calculations

Groundwater travel times are calculated using Darcy's law and interpreting flow pathways through geologic cross sections, which are constructed from well logs in the study area. Since the location, thickness, and number of clay layers varies at each well location, the continuity of any given layer is often poorly known. These cross sections are then used to project flow paths of recharged water from the spreading basins to each well. Darcy's law is applied to these data to calculate travel times to each well



after estimating local values of hydraulic conductivity and porosity. In the current study, data for the hydrogeologic calculations were obtained from Bookman-Edmonston Engineering, Inc. (1994).

### Groundwater Dating

Determining residence time for shallow groundwater is also possible using environmental tracers such as  $T/{}^3\text{He}$  (Schlosser et al. 1989; Cook and Solomon 1997). Most of the tritium, a radioactive isotope of hydrogen currently in the environment, was released to the atmosphere in the late 1950s and early 1960s during above ground nuclear bomb tests. The tritium concentration in precipitation reached a maximum in the mid-1960s, 2–3 orders of magnitude higher than natural levels. Since then, its concentration has decreased quasi-exponentially. The age of the groundwater can be calculated using the radioactive decay relationship between tritium and its daughter isotope,  ${}^3\text{He}$

$$t = \frac{t_{1/2}}{\ln(2)} \left( 1 + \frac{[{}^3\text{He}]_{\text{tri}}}{[\text{T}]} \right) \quad (1)$$

where  $t$ =time (apparent groundwater age);  $t_{1/2}$ =half-life of tritium (12.43 years);  $[\text{T}]$ =measured tritium content; and  $[{}^3\text{He}]_{\text{tri}}$ =concentration of  ${}^3\text{He}$  derived from the decay of tritium.  $[{}^3\text{He}]_{\text{tri}}$  is calculated from a He mass balance that considers other He sources (Schlosser et al. 1989; Cook and Solomon 1997). In addition to tritium decay, He sources in the subsurface include He from atmospheric equilibrium, excess air,  $\alpha$ -decay of U and Th series nuclides and their associated production of  ${}^3\text{He}$ , and the mantle (Schlosser et al. 1989; Cook and Solomon 1997). For shallow groundwater, the contribution of  ${}^3\text{He}$  from U and Th decay, and the mantle is small and generally negligible unless  ${}^4\text{He}$  concentrations are elevated ( $>1.25 \times 10^{-7}$  cm standard temperature and pressure (STP)/g). The typical analytical uncertainty of this method is  $\pm 2$  years. However, the uncertainty of the apparent age can be much larger if mixing between flow paths of different ages occurs. Because  $T/{}^3\text{He}$  ages do not mix linearly—the mixed age is weighted by each flow path's initial tritium content—this technique leads to overestimation of groundwater age when mixing occurs with young water ( $<50$  years) and an underestimation of groundwater age when a portion is older than 50 years.

For this study,  $T/{}^3\text{He}$  samples were collected independently from the  $\text{SF}_6$  experiment as part of the State of California's Groundwater Ambient Monitoring and Assessment (GAMA) program and were obtained from G. Bryant Hudson at Lawrence Livermore National Laboratory (LLNL). The GAMA program is sponsored by the California State Water Resources Control Board, and is aimed at assessing the susceptibility of groundwater to contamination (California SWRCB 2006).

### $\text{SF}_6$ Tracer Experiment Methods

From February 14 to 21, 2003, 99.99% pure  $\text{SF}_6$  gas was injected into recharge water contained in 23 spreading basins and the San Gabriel River at the Montebello Forebay Spreading Grounds. The tracer was injected by bubbling for about 30 min through a diffusion stone located near each basin's edge at a rate of about 40cc/min every 1–3 days. At the start of the injection period, all of the basins and the San Gabriel River above inflatable dam No. 5 were full following an unusually large February rainfall event. We estimate that more than  $10 \times 10^6 \text{ m}^3$  of water were tagged with tracer.

Surface water samples (5–8 samples/pond) were collected in 15-mL Vacutainers from a small boat 1–2 days after each injection event. Collection of surface water samples continued after the injection period until March 18, 2003 when the surface water concentration reached approximately zero. Personnel from the Water Replenishment District of Southern California collected well samples in 15-mL preweighed Vacutainers every 2–10 weeks after the injection period from ten monitoring and 18 production wells for a period of 2 years (Table 1).

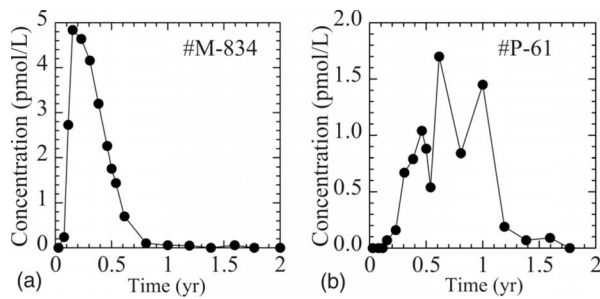
All samples were shipped to the University of California, Santa Barbara for analysis following the procedures outlined by Clark et al. (2004). In the laboratory, the Vacutainers were weighed to determine the sample size (typically between 5 and 10 mL) and carefully filled with ultrahigh purity nitrogen gas until the final pressure was equal to about 1 atm. A known volume ( $\sim 1.5$  mL) of headspace gas was transferred to a gas chromatograph (GC) equipped with an electron capture detector (ECD).  $\text{SF}_6$  was separated from other gases with a Molecular Sieve 5a column held at room temperature. During the GC runs,  $\text{SF}_6$  eluted before air at 0.7–0.9 min depending on carrier gas flow conditions. The GC detector response was calibrated about every ten samples with standards [ $\sim 148$ ,  $\sim 524$ , and  $\sim 1,947$  parts per trillion by volume (pptv)] prepared by Scott-Marrin, Inc. (Riverside, Calif.). The precision and detection limits of this method are  $\pm 5\%$  and 0.05 pmol/L, respectively.

Vacutainers are convenient collection and reaction vials, but problems can arise when dissolved concentrations are  $<0.2$  pmol/L and have head space concentrations  $<10$  pptv. Any leak or improper collection technique that allows air to be collected along with water will result in detectable  $\text{SF}_6$  because its mixing ratios in urban air are variable and can exceed background value ( $\sim 5$  pptv) by several orders of magnitude (Ho and Schlosser 2000). This variability is due to the commercial use of  $\text{SF}_6$  in the electric power industry. To eliminate the possibility of false positive detections of tracer, a strict set of criteria were established to identify detection at a well.  $\text{SF}_6$  must be detected in at least two sequential samples, and have a concentration greater than or equal to 0.1 pmol/L (twice the detection limit) during at least one sampling event. A false positive detection can be due to a leak in the Vacutainer, allowing for air contamination of the sample or incomplete flushing of the sampling tube and needle, which results in the collection of small air bubbles previously contained in the sampling equipment. There were 494 well samples analyzed during this study, with 6% false positive detections (28 samples) and 14% true detections (67 samples).

### Results and Discussion

Concentration of  $\text{SF}_6$  within the spreading basins (ponds) varied between 4 and 68 pmol/L with an average concentration of 30 pmol/L [all concentration data can be found in McDermott (2006)]. Following each injection, concentrations in the ponds decreased quasi-exponentially with a time scale of about 1 week due to gas loss and dilution with the inflow of nontagged source water. Variability was sometimes higher within ponds than between ponds. Various factors influenced this temporal and spatial variability. Individual pond wetted area, percolation rate, and basin mean depth varied, as did time since the last injection and the water inflow rates. Finally, the amount of  $\text{SF}_6$  injected was the same at each pond despite their size differences.

$\text{SF}_6$  was not detected in samples collected 6 and 2 months prior to the start of the tracer experiment. During the 2 years



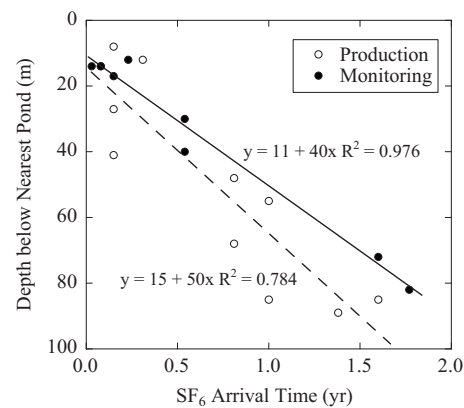
**Fig. 2.** SF<sub>6</sub> breakthrough curves at monitoring well No. M-834 and production well No. P-61

following the tracer release, nine of the ten monitoring wells sampled received SF<sub>6</sub>, indicating successful transfer through the unsaturated zone to the water table (Table 1). Breakthrough curves are characterized by tracer first arrival, peak, center of tracer mass, and tailing. The first arrival is the best indicator of preferential flow, which cannot be determined to a high degree of certainty with either geochemical dating techniques or numerical flow modeling. Yet, this is the most important time scale when evaluating potential transport of reactive contaminants near MAR facilities. Wells located adjacent to one spreading basin [Fig. 2(a)] generally show only one peak, while wells located adjacent to multiple spreading basins [Fig. 2(b)] have multiple breakthrough peaks, indicating the multiple pathways leading to wells. Many of the well breakthrough curves show tailing [Fig. 2(a)]. Tailing could be the result of relatively small volumes of SF<sub>6</sub> tagged water reaching the well through slower (deeper) flow paths (Clark et al. 2005). It could also be due to retardation from trapped gas (Fry et al. 1995; Donaldson et al. 1997; Heilweil et al. 2004; Clark et al. 2005).

All monitoring wells where tracer was detected were located within 150 m of the spreading basins. Those with detections within 6 months have screen depths <15 m, and all those showing later detection have depths of <70 m below the nearest spreading basin. At the one shallow monitoring well (No. M-832), which was more than 1 km from the spreading grounds, SF<sub>6</sub> was not detected. There is a strong correlation between vertical depth to perforation and travel time for all monitoring wells sampled (Fig. 3). There is no relationship between horizontal distance and travel time.

SF<sub>6</sub> was detected at 11 of the 18 production wells (Table 1). Four of the wells showed tracer arrival within 6 months (Nos. P-55, P-58, P-61, and P-65). These wells have perforation depths of <40 m below the nearest spreading basin and are considered to be in an unconfined aquifer based on hydrostratigraphy constraints (Bookman-Edmonston, 1994). All other wells, regardless of depth, are considered to be in a confined or semiconfined aquifer. As in the monitoring wells, there is a relationship between tracer arrival time and depth to the top of the perforation (Fig. 3) and no correlation between horizontal distance and travel time. Depth may be the most important factor influencing travel time because the deeper the well perforation is, the more likely is it for the screen to be situated below layers with low hydraulic conductivity. Since horizontal hydraulic conductivities are generally an order of magnitude greater than vertical hydraulic conductivities, the rate-limiting factor for travel time is the vertical hydraulic conductivity.

Previous work by Clark et al. (2004) using only monitoring wells at an artificial recharge facility found a relationship between



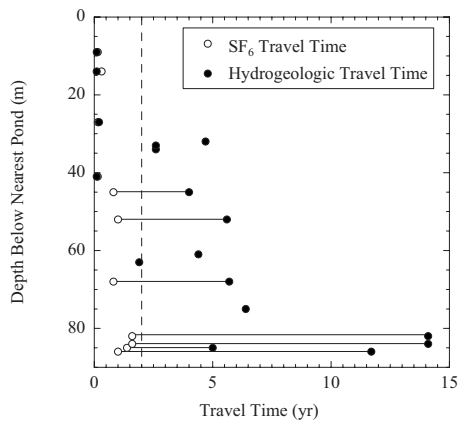
**Fig. 3.** SF<sub>6</sub> initial arrival times with vertical depth for all production and monitoring wells. Nonzero intercept of ~10–15 m is expected due to local water table levels.

travel time and both horizontal distance and depth. The lack of relationship between travel time and distance in this study most likely results from the experimental design: only wells within 150 m of the spreading ponds were sampled.

Pumping rate is a poor predictor of travel time to the production wells. However, for wells screened at the same depth, production wells have faster arrival times than monitoring wells. This indicates that although production is not the primary factor, it does have an influence. Groundwater pumping combined with the fact that monitoring wells are, on average, located closer to the edge of spreading ponds than production wells, contribute to the lower regression coefficient.

The maximum groundwater SF<sub>6</sub> concentration was 4.8 pmol/L at monitoring well No. M-834, 15% of the mean concentration in the surface water, and 20% of the concentration in the nearest spreading basin. The maximum concentration observed at each production well was much lower than at the monitoring wells. At seven of the production wells with SF<sub>6</sub> detections, the maximum concentrations were <0.4 pmol/L, an order of magnitude less than at the monitoring wells and two orders of magnitude less than the spreading basins. These lower concentrations are most likely due to dilution of the tagged water with untagged groundwater caused by the long screened intervals. However, for wells with detection, there is no discernible effect of screen length on maximum concentration, indicating that dilution factors are not dependent on screen length. Using the average pond concentration of 30 pmol/L, and assuming dispersion reduces the concentration to a maximum of 5 pmol/L (obtained from maximum peak detected at monitoring well No. M-834), detection is possible with a mixture of 2% tagged/98% untagged water if sampling occurred when the maximum groundwater concentration reached the well. Sampling during nonpeak times would require a larger fraction of tagged water.

Bookman-Edmonston's (1994) hydrogeologic analysis divided the production wells into two groups. The first consisted of four shallow wells (Nos. P-55, P-58, P-61, P-65) having very short mean travel times of <0.2 years (Table 1) and screened in the unconfined aquifer. In the present study, the hydrogeologic mean travel times and SF<sub>6</sub> initial arrival times for these wells are consistent, both indicating travel times of 0.3 years (16 weeks) or less. The hydrogeologic study identified all other wells as having perforations in a confined aquifer with little direct contact with surface water. At these wells, estimated mean travel times range between 2.5 and 14 years (Table 1).

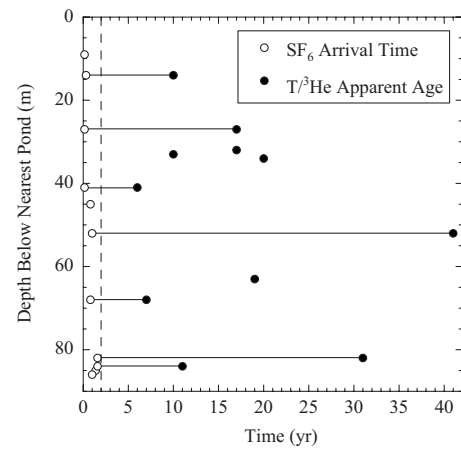


**Fig. 4.** Comparison of SF<sub>6</sub> initial arrival times and hydrogeologic travel times (Bookman–Edmonston 1994). Horizontal lines connect each well's SF<sub>6</sub> and hydrogeologic travel times. Vertical dashed line represents end of 2 year-long SF<sub>6</sub> experiment. For wells with SF<sub>6</sub> travel times greater than 2 years, only hydrogeologic travel times are plotted.

During the 2-year SF<sub>6</sub> experiment, four production wells near the Rio Hondo spreading grounds (Nos. P-62, P-88, P-89, P-99), and three production wells near the San Gabriel spreading basins (Nos. P-11, P-12, P-18), which were determined to be in the confined aquifer, received tracer (Fig. 4). The deliberate tracer experiment allows for analysis of two travel times: initial and mean. In this study, initial arrival times are used and compared to calculated hydrogeologic mean travel times. Although there is a time lag of a few months between initial and mean arrival times, the comparison is valid as the inconsistency between the two methods for the confined aquifer is on the order of years. This comparison between the tracer and hydrogeologic travel times suggests that discontinuities (gaps, fractures, or interbedded layers of coarse material) within the confining layers exist. An alternate possibility for the faster than expected arrival of SF<sub>6</sub> is vertical leakage along the production well seals allowing water from the upper aquifers to be pulled out of the deeper perforations. While there is no direct evidence for or against this occurrence, the strong relationship with depth seen at the production wells suggests that leakage is not a significant problem. Within the time constraints of the 2-year tracer experiment, analysis of the accuracy of the hydrogeologic travel times to all of the production wells could not be completed.

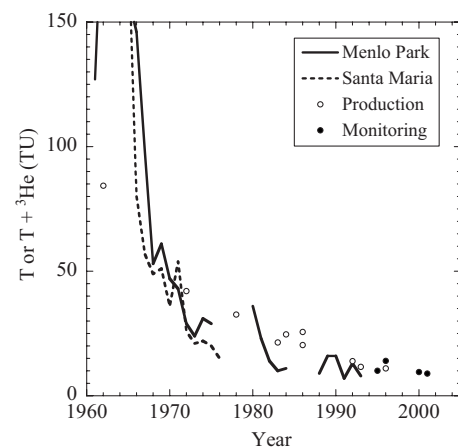
The T/<sup>3</sup>He apparent ages at production wells, which were collected independently of the SF<sub>6</sub> tracer experiment, were between 7 and 40 years, all significantly older than travel times determined by the other two methods (Fig. 5; Table 1). The older apparent ages reflect mixing of young and old groundwater within the well. When mixing of water of different ages occurs, this technique leads to differences between the T/<sup>3</sup>He apparent age and mean groundwater age because the mixed age is weighted by each flow path's initial tritium content. Good agreement between travel times determined with deliberate tracer experiments and T/<sup>3</sup>He ages have been found at monitoring wells elsewhere (Clark et al. 2004).

The T/<sup>3</sup>He apparent ages were plotted against each well's initial tritium concentration to determine what percentage of water was recharged prior to ~1950 and what percentage was recharged after ~1950 (referred to as modern water) following the method described by Manning et al. (2005). Annual average



**Fig. 5.** Comparison of SF<sub>6</sub> initial arrival times and T/<sup>3</sup>He apparent ages, which were obtained from G. Bryant Hudson at LLNL. Horizontal lines connect data from same well. For wells with SF<sub>6</sub> travel times greater than 2 years, only T/<sup>3</sup>He apparent ages are plotted.

tritium concentrations in precipitation for 1961–2000 were obtained from the Menlo Park and Santa Maria records (IAEA 2006). The initial tritium concentrations for all wells sampled were plotted along with the precipitation data (Fig. 6). Samples that fall on or above the curve fit line probably contain >80% modern water, while samples that fall below the line probably contain a component of water recharged prebomb. One sample (well No. P-18) plots below the curve fit line, indicating mixing with prebomb water. The T/<sup>3</sup>He apparent age of 41 years for this well also indicates prebomb water mixing since it is only possible to obtain an age greater than 40 years if there is a large component of prebomb water and only a small fraction of modern water (Table 2). Further evidence for a large prebomb component comes from the elevated <sup>4</sup>He concentration found in this sample ( $1.25 \times 10^{-7}$  cm STP/g). Given typical groundwater <sup>4</sup>He accumulation rates, <sup>4</sup>He concentrations this high would require a residence time of significantly greater than 50 years to occur (Manning et al. 2005). All other samples plot on or above the curve fit line for tritium concentrations in precipitation, indicating



**Fig. 6.** Plot of initial T (T + <sup>3</sup>He) content of well samples with annual average T content in precipitation for coastal sites in California. Precipitation data are from International Atomic Energy Agency.



**Table 2.** Hypothetical Mixtures of Different Age Flow Lines and Resulting T/<sup>3</sup>He Apparent Age; Prebomb Water Is Assumed to Have Recharged in 1940s and Recent Water Recharged in 2000

	Water fraction					
Prebomb	0	0	0	0	0	95
1960s	0	5	10	20	45	0
1970s	10	5	10	20	45	0
1980s	10	0	0	0	0	0
Recent	80	90	80	60	10	5
Apparent age	<b>7</b>	<b>11</b>	<b>18</b>	<b>25</b>	<b>33</b>	<b>43</b>

they contain >80% modern water. Since MAR operations at Montebello began before 1960, the majority of the pumped water may have been artificially recharged at this site.

One of the major conclusions of the SF<sub>6</sub> experiment was the finding that arrival time is a strong function of depth. To test this finding and determine if a potential strategy of deepening wells will lead to compliance with the CDPH reuse rules, a second experiment was conducted in 2005. During this experiment, one production well (No. P-61) was chosen for modification; a packer was placed within the well at 62 m to isolate the deep-screened portion of the well from the shallow-screened region, thus increasing the depth to the top of the screen from 42 to 68 m. Well No. P-61 was ideal for the experiment because the well has a long screened interval which extended from the unconfined to the deeper semiconfined aquifers, and had very fast tracer arrival (0.15 year) in the initial study. The second injection was completed from June 14 to 26, 2005 using the same procedure used in 2003. The only differences were fewer ponds were wet and actively recharging the groundwater system, and the pond percolation rate was lower. SF<sub>6</sub> was injected into ponds nearest to well No. P-61 (Nos. 1–5) at the Rio Hondo site. These ponds were initially full and were partially refilled once during the injection phase. In addition to above and below the packer in well No. P-61, samples were collected from four monitoring wells and one other production wells every 2–4 four weeks for a total of 1 year.

The modified well, No. P-61, two monitoring wells (Nos. M-830 and M-834), and one production well (No. P-65) were sampled during both tracer experiments. The SF<sub>6</sub> travel times to the unmodified wells were identical (within sampling error) in both studies, indicating no significant change in the flow system occurred between events. At well No. P-61, SF<sub>6</sub> was detected simultaneously, at 0.5 year (26 weeks), above and below the packer. This arrival time is significantly longer than the 0.15 year (8 weeks) detection observed during the 2003 experiment confirming that the well modifications with the packer successfully increased the travel time. The concurrent detection, along with approximately equal hydraulic head values above and below the packer, indicate a hydraulic connection and potential movement of water between the two regions. Regardless of the potential exchange, in the 2005 experiment the travel time to both regions was 6 months, and therefore compliant with the CDPH draft rules.

## Conclusions

Previous work examining groundwater residence time with monitoring wells found a relationship with depth and horizontal distance, and saw good agreement between multiple tracers (Clark et al. 2004). The current study used three complementary methods

to examine groundwater residence time to wells within 150 m of spreading basins, the distance specified by the CDPH reuse regulations. In contrast with the earlier study, this study found that SF<sub>6</sub> initial arrivals are a function of depth, not horizontal distance. Results also show that travel times to production wells typically are shorter than travel times to monitoring wells after normalizing for depth differences. The importance of depth was clearly illustrated during the second deliberate tracer experiment when a production well was modified to deepen its intake zone. With a deeper screen, the travel time increased significantly from 0.15 to 0.5 year.

The CDPH draft regulations (CDPH 2007) call for a minimum residence time of 6 months for groundwater containing a component of recharged reclaimed water in order to allow for virus inactivation. The draft regulations also indicate a minimum extraction distance of 150 m from the point of infiltration; this measure is aimed at ensuring the 6-month residence time. This study suggests that using horizontal distances does not ensure a specified travel time within the groundwater system; the depth of production is a better criterion to ensure desired residence time close to spreading ponds.

Comparison of the SF<sub>6</sub> and hydrogeologic calculations found that SF<sub>6</sub> initial arrivals and calculated hydrogeologic mean travel times agree at production wells in the unconfined aquifer, but in the (semi-) confined aquifers some SF<sub>6</sub> initial arrival times are significantly shorter than hydrogeologic mean travel times. This discrepancy indicates that SF<sub>6</sub> is transported along preferential flow paths through discontinuities in the clay layers or “leaky” confining units. T/<sup>3</sup>He apparent ages are much older than SF<sub>6</sub> travel times, indicating that production wells draw in both young (<2 years) and old (>10 years) water components.

Multiple methods (SF<sub>6</sub> tracer experiment, hydrogeologic calculations, geochemical dating) were compared and used to construct a more robust model of flow and transport to production wells near the Montebello MAR facility. Quantification of the fastest flow paths with a deliberate SF<sub>6</sub> tracer experiment was possible, but limited by the duration of the experiment, which in this case was 2 years. Identifying and quantifying travel times of these fast paths is critical for regulating MAR operations. The concurrent use of a deliberate tracer experiment and hydrogeologic calculations, which are based on geologic information obtained from well logs, validates and leads to better models of the hydrostratigraphy. In production well environments where significant mixing of groundwater from multiple flow paths occurs due to the long screen (intake zone), T/<sup>3</sup>He apparent ages and SF<sub>6</sub> results should differ. In this study, a comparison of these two methods determined that many of the production wells are drawing in an old component with a mean apparent age of decades as well as very recent water (<2 years) recharged from the spreading basins.

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