Application of Slug Tests to Detect Reduced Aquifer Permeability Due to Byproducts of Bioremediation of Groundwater Contaminants at the Naval Air Warfare Center, West Trenton, New Jersey

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Abstract

Seven wells previously used in a biostimulation injection to enhance the biotic dechlorination of trichloroethylene in a fractured sedimentary rock aquifer were slug tested to identify possible transmissivity decreases. Aquifer permeability may reduce as products of enhanced bacterial activity such as biomass and mineral precipitates develop in a well and accumulate in fractures. Such aquifer clogging can cause a transmissivity decrease that may be observable with slug tests. Comparisons to pre-biostimulation slug tests of five of these wells indicate a transmissivity decrease of at least two orders-of-magnitude at one minimally fractured well and negligible or no transmissivity change at the other wells. The large decrease of T coincides with increased sulfate-reducing groundwater conditions in that well following biostimulation, which may indicate a possible upsurge of sulfate-reducing bacteria activity.

Introduction

The former Naval Air Warfare Center (NAWC) in West Trenton, New Jersey, was a U.S. Navy jet engine testing facility in operation from the 1950s to the mid-1990s, and is presently a field site for U.S. Geological Survey (USGS) Toxic Substances Hydrology Program research on groundwater contamination in fractured sedimentary rock. Trichloroethylene (TCE), which is toxic to humans at concentrations greater than 0.005 mg/l (EPA), was leaked into the subsurface during industrial operations at NAWC and has been identified at concentrations over 10,000 mg/l in the fractured Newark Basin mudstones and shales that underlie the site (fig. 1, 2) (Lacombe, 2000). A series of biostimulation injections performed by Geosyntec Consultants™ at NAWC in 2005 and 2008 (deFlaun et al., 2006; deFlaun, unpublished) utilized emulsified vegetable oil (EVO) as an electron donor to stimulate biotic dechlorination of TCE and daughter products cis-1,2 dichloroethylene (DCE) and vinyl chloride (VC).

The two biostimulation injections may have induced biofouling in wells used during the study. Biofouling, as it relates to groundwater hydrology, is a change of physical and/or chemical characteristics of an aquifer due to growth of microorganisms (Cullimore, 1999). Such effects are commonly observed in contaminated aquifers undergoing in-situ bioremediation. Nutrient-rich biostimulants such as EVO are injected into the subsurface to enhance bacterial activity, which can cause biofouling when biomass growth and/or precipitation of mineral byproducts accumulate in the aquifer and reduce hydraulic properties such as transmissivity (Barnes and Clarke, 1969; Walter, 1997; MacDonald et al., 1999; Ross et al., 2001; Houben, 2003; Lutes et al., 2003; Castegnier et al., 2006; Hoelen et al., 2006; Seifert and Engesgaard, 2007; Larroque and Franceschi, 2011; Page et al., 2011; Smith and Roychoudhury, 2013). In fractured rock aquifers, biofouling and subsequent decreases of T will occur within the fractures themselves, which act as the primary conduits of groundwater flow through the aquifer. Fractured rock aquifers are notoriously heterogeneous, the Newark Basin being no exception (Morin et al., 1997; Lacombe, 2000; Fan et al., 2007; Herman, 2010; Michalski, 2010; Fiore, in press), so the extent of biofouling due to injection of biostimulants is likely localized to the volume of aquifer immediately surrounding the well. Slug tests, a common method of estimating hydraulic properties, are most effective in estimating T in the immediate vicinity of a well, so localized T decreases associated with biofouling should be observable with slug tests. The groundwater released during slug tests will predominantly originate from fractures intersecting the well (Barker and Black, 1983). Clogging will decrease the cross-sectional area of the conduit, and the slug test will take longer to recover to static water levels, underestimating T. Decreases of hydraulic properties such as T due to buildup of biofouling materials in fractures have been observed in biostimulated fractured sedimentary rock (Page et al., 2011) as well as limestone aquifers (Ross et al., 2001; Castegnier et al., 2006), but no studies have used slug testing to detect such decreases. Therefore, the objective for this project was to implement slug tests to determine if biofouling has altered the T at wells involved in biostimulation injections at NAWC in 2005 and 2008, and to pose possible explanations for any indications of biofouling.

Site Hydrogeology

Detailed hydrogeologic descriptions of NAWC are found in Lacombe (2000) and Lacombe and Burton (2010). NAWC is situated on the Lockatong Formation and Stockton Formation of the Newark Basin (fig. 2). A thrust fault separates the two formations (figs. 2, 3), which is believed to act as a barrier to groundwater flow. Most groundwater contamination occurred in the Lockatong Formation, which underlies a larger portion of the site, while the Stockton Formation is largely uncontaminated (fig. 1). The Lockatong at NAWC is primarily fractured mudstone and shale and the Stockton is primarily fractured sandstone (fig. 2). Preferential groundwater flow-paths are through fractures, rather than the primary porosity of the rocks. General groundwater flow direction is along the strike of the beds toward the southwest and down-dip toward the northwest, as evidenced by migration of the TCE plume relative to the daughter DCE plume (fig. 1) (Lacombe, 2000; Lewis-Brown and Rice, 2002). The biostimulation injection occurred up-strike and up-dip from the primary contaminant plume. Since the 1990s, a pump-and-treat system has been in use for groundwater remediation and to prevent TCE from entering the nearby Gold Run stream that discharges to the Delaware River (Lacombe, 2000; Lewis-Brown and Rice, 2002). The pump-and-treat system has been only slightly successful in removing TCE and daughter products (Lacombe, 2011), so alternative remediation methods such as these biostimulation injections were necessary to facilitate TCE removal.

Materials and Methods

In 2012, slug tests were performed on 7 wells used in the biostimulation injections to obtain T estimates: well names 16BR, 17BR, 30BR, 38BR, 41BR, 74BR, and BRP-1 (fig. 3). Wells 47BR and 68BR were also used in the biostimulation (fig. 3), but are not included in this report. 47BR is open to unconfined regolith with a differing flow scheme (Lacombe, 2000; Tiedeman et al., 2010). 68BR was unavailable to be slug tested at the time. The tests deployed mechanical slugs that raise the water level when rapidly inserted down a well (“slug in” test) and lower the water level when rapidly removed from a well (“slug out” test) (fig. 4). Pressure transducers placed down the borehole (fig. 4) recorded water-level data at 0.25-60 second (sec) intervals. Wells with faster water-level recovery used shorter intervals, and wells with slower recovery used longer intervals. Water levels were also manually sounded with an electric M-scope to calibrate transducer data and convert from millivolts (mV) to water-level depth in meters (m).

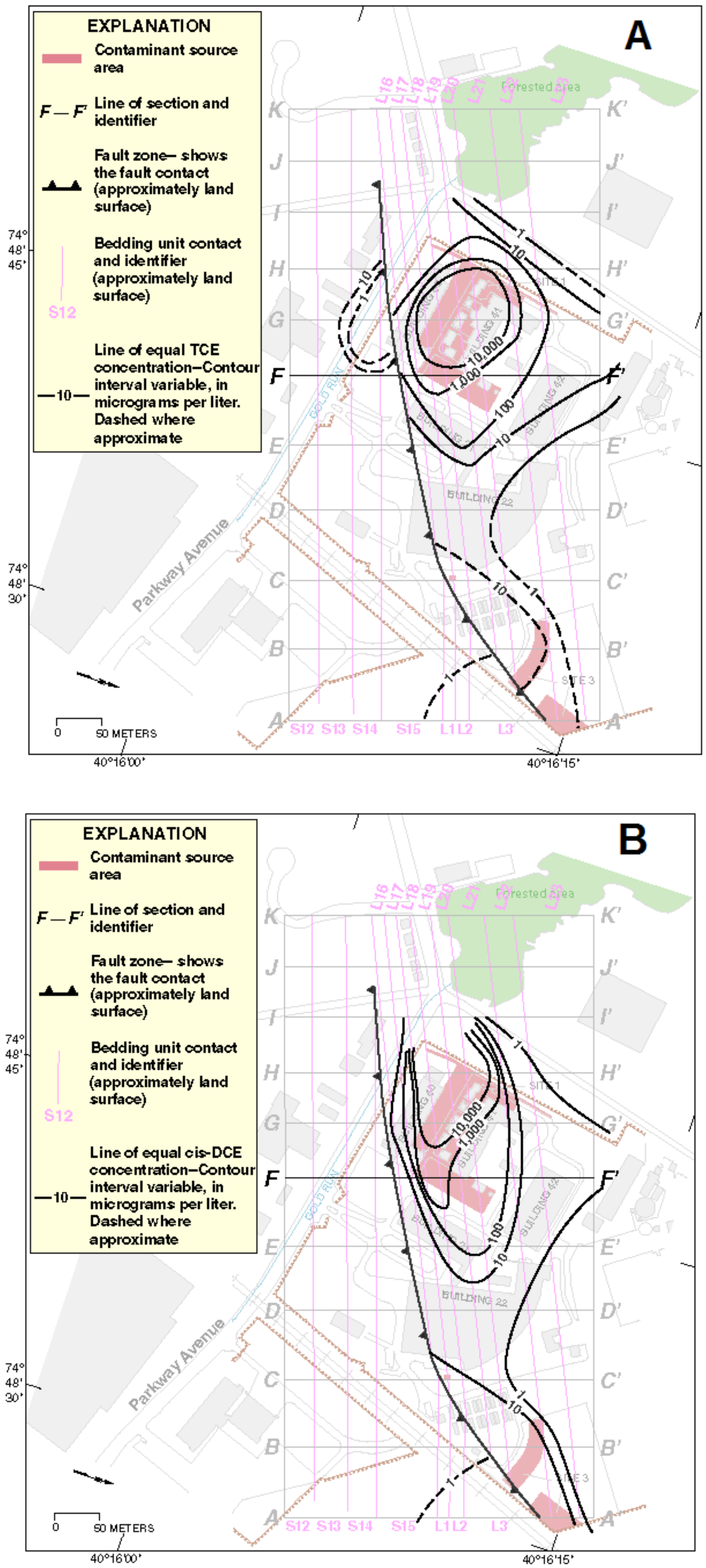


Figure 1. (a) Map of TCE concentrations at the Naval Air Warfare Center site. (b) Map of DCE concentrations.

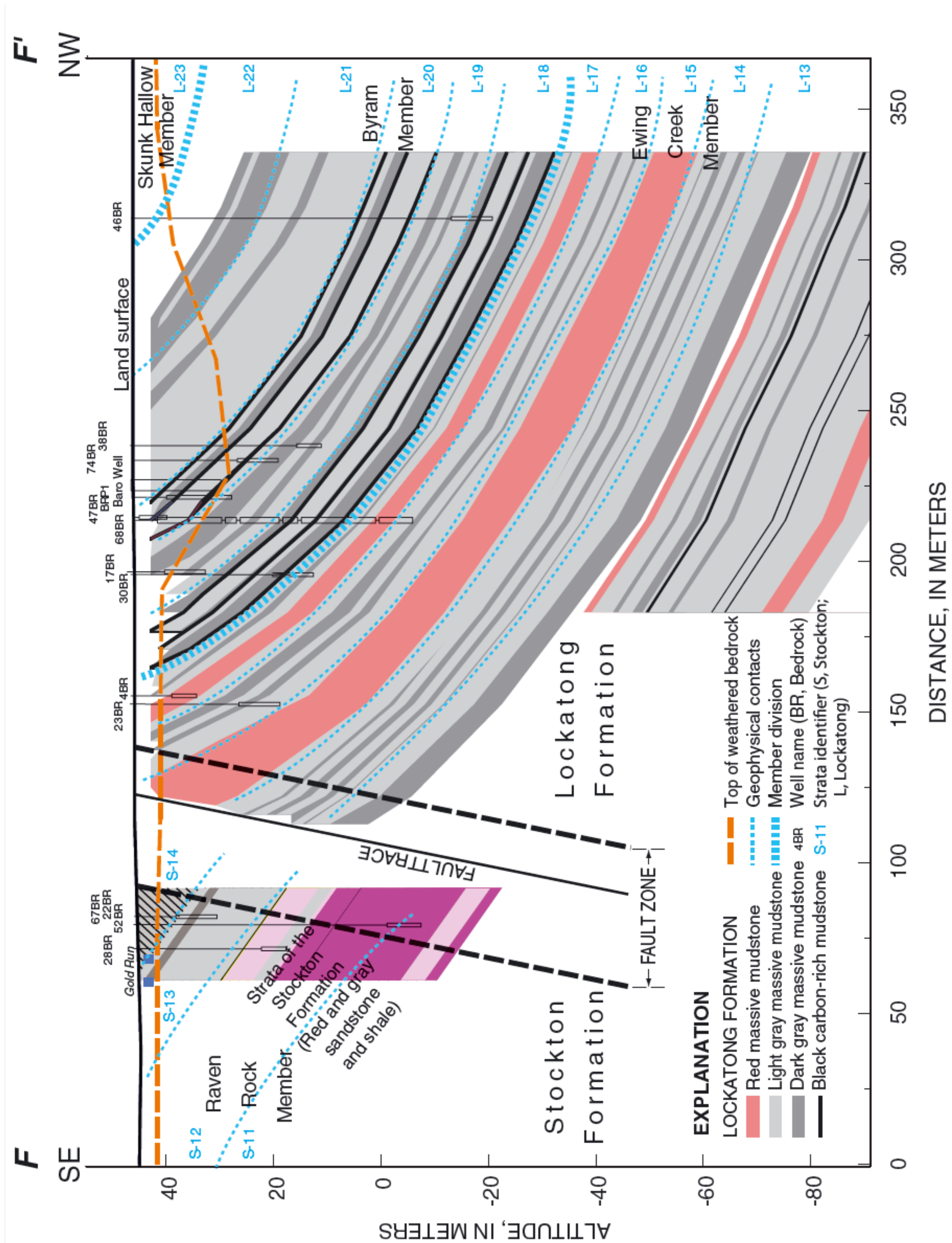
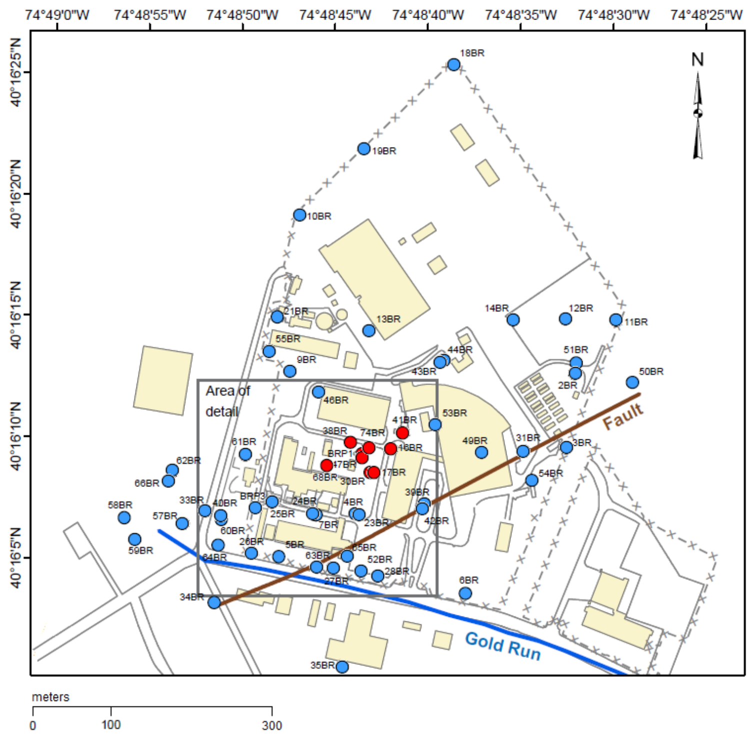


Figure 2. Cross section F-F’ (see fig. 1) of Naval Air Warfare Center site. From Lacombe and Burton (2010).



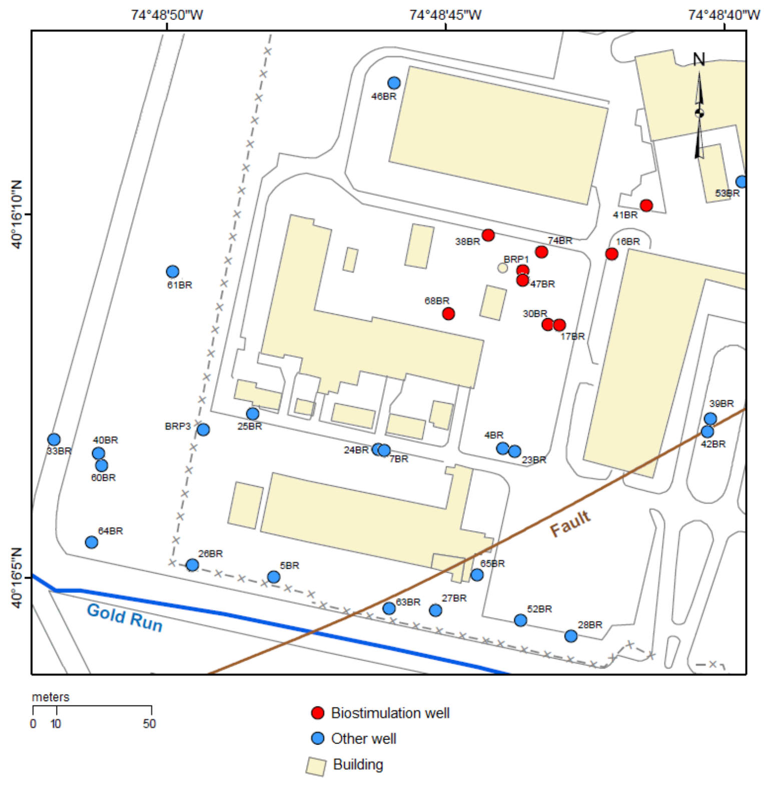


Figure 3. (top) Map of Naval Air Warfare Center site with well locations. (bottom) Map detail.

Slug removal data were analyzed over slug insertion data because slug removal causes less splashing than slug insertion and provides a “cleaner” initial water-level displacement. The water-level displacement from slug tests and the recovery time to static levels can provide a measure of groundwater flow through the aquifer (fig. 5a). The solution of Cooper-Bredehoeft-Papadopulos (Cooper et al., 1967) for slug tests of confined, non-leaky aquifers was the analytical method of choice for these tests. The non-leaky condition is valid because groundwater released during slug removal emanates primarily from fractures and there is negligible leakage between bedrock porosity and fractures during slug tests. The Cooper et al. (1967) solution assumes a homogeneous, isotropic, and isopachous aquifer, which is likely applicable over the very small volume of aquifer affected by slug tests. The solution also assumes full well penetration of the aquifer. Since groundwater flow occurs in fractures, a confining unit of competent bedrock is likely positioned at the top and bottom of each well screen and this assumption is applicable. Additional Cooper et al. (1967) assumptions (horizontal potentiometric surface, unsteady flow, instantaneous release from storage, etc.) are also considered valid. These conditions are rarely if ever encountered by any real aquifer in nature, but provide reasonable approximations of hydraulic properties of fractured rock (Shapiro and Hsieh, 1998). The Cooper et al. (1967) analysis requires plotting normalized water-level displacement over time, matching the graphed slug test data to a type curve (fig. 5b, 6). The shape of the type curve is controlled by a specified storage coefficient, as well as radii of the casing and well screen (fig. 5b). Type curves were generated and T estimates were calculated using the software AQTESOLV (Duffield, 2000).

Estimates of hydraulic conductivity (K) are available from slug tests of NAWC wells performed in 1997 (Lewis-Brown and Rice, 2002), prior to any biostimulation event. For consistency, K estimates from 1997 were converted to T by multiplying by the saturated thickness (b) specified in Lewis-Brown and Rice (2002). T was preferred over K to avoid any possible inaccuracies of b. Both studies utilized the Cooper et al. (1967) solution for data analysis, which provides a T estimate without requiring a specified b. Comparison of T estimates from 1997 and 2012 may indicate clogging of the fractures due to biofouling if 2012 estimates are less than 1997 estimates.

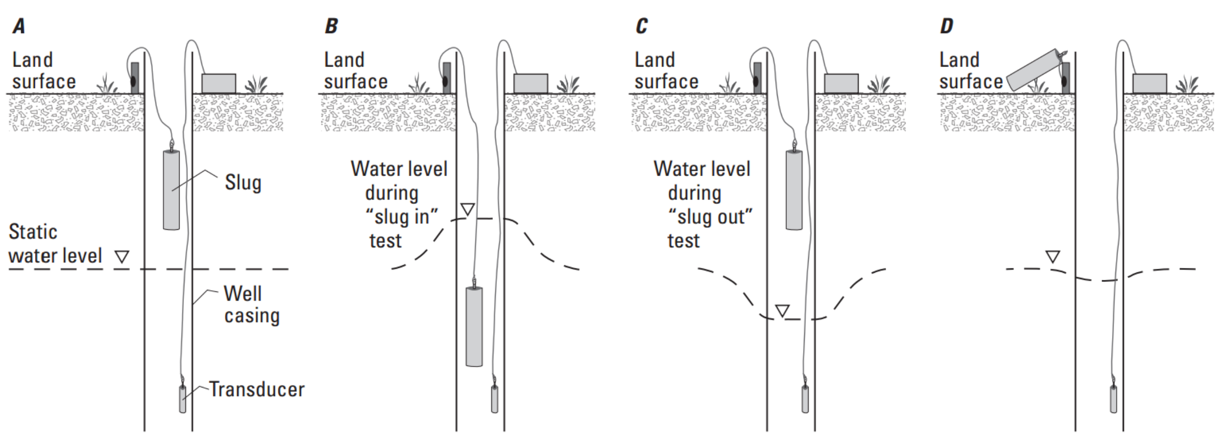


Figure 4. Well diagram with solid-body slug (a) positioned above water level for slug in test, (b) submerged below water level for slug in test, (c) removed just above water level for slug out test, and (d) removed from well for slug out test. Modified from Cunningham and Schalk (2011).

Table 1. Comparison of 1997 and 2012 transmissivity (T) estimates at biostimulation wells. Data reported to one significant figure. *Transmissivity too low to measure with slug tests, value is less than lowest measured T estimate.

Well Name	1997 T (m^2/day)	2012 T (m^2/day)
16BR	6	3
17BR	50	40
30BR	0.6	<0.007*
38BR	<0.009*	<0.007*
41BR	40	9
74BR	---	2
BRP-1	40	40

Results

Water-level displacement data exhibited a reasonable fit to Cooper et al. (1967) type curves (fig. 6). Comparison of slug test estimates of T from 1997 and 2012 are listed in table 1. Wells 16BR, 17BR, 30BR, and 41BR each decreased in T over the 15-year time span. Well BRP-1 indicated no T change. Wells 38BR and 74BR had an unknown change of T. Water levels in 38BR showed no indication of recovery during testing, which indicates a very slow recovery and that T is too low to measure using slug tests. Too-slow water-level recovery also occurred during the 2012 slug test of 30BR. The smallest measured T estimate from slug tests at any NAWC well is 0.007 m^2/day (Fiore, in press), so the 2012 T at 30BR and 38BR is most likely less than this estimate. The smallest measured T estimate by Lewis-Brown and Rice (2002) was 0.009 m^2/day, so T at 38BR in 1997 is most likely <0.009 m^2/day. Well 74BR did not exist in 1997, so no prior slug test data is available for that well.

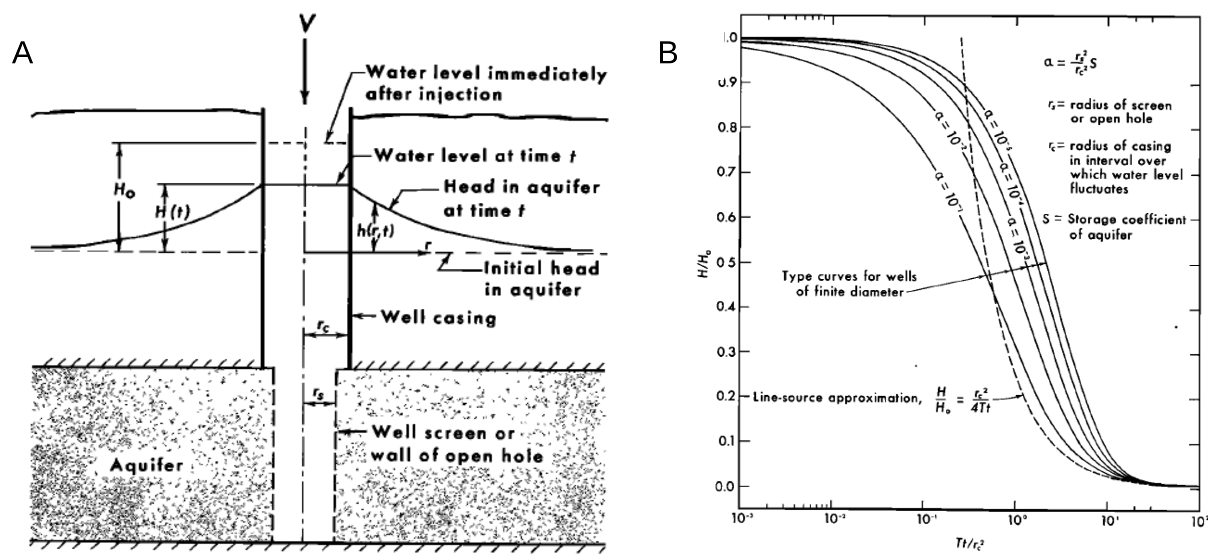
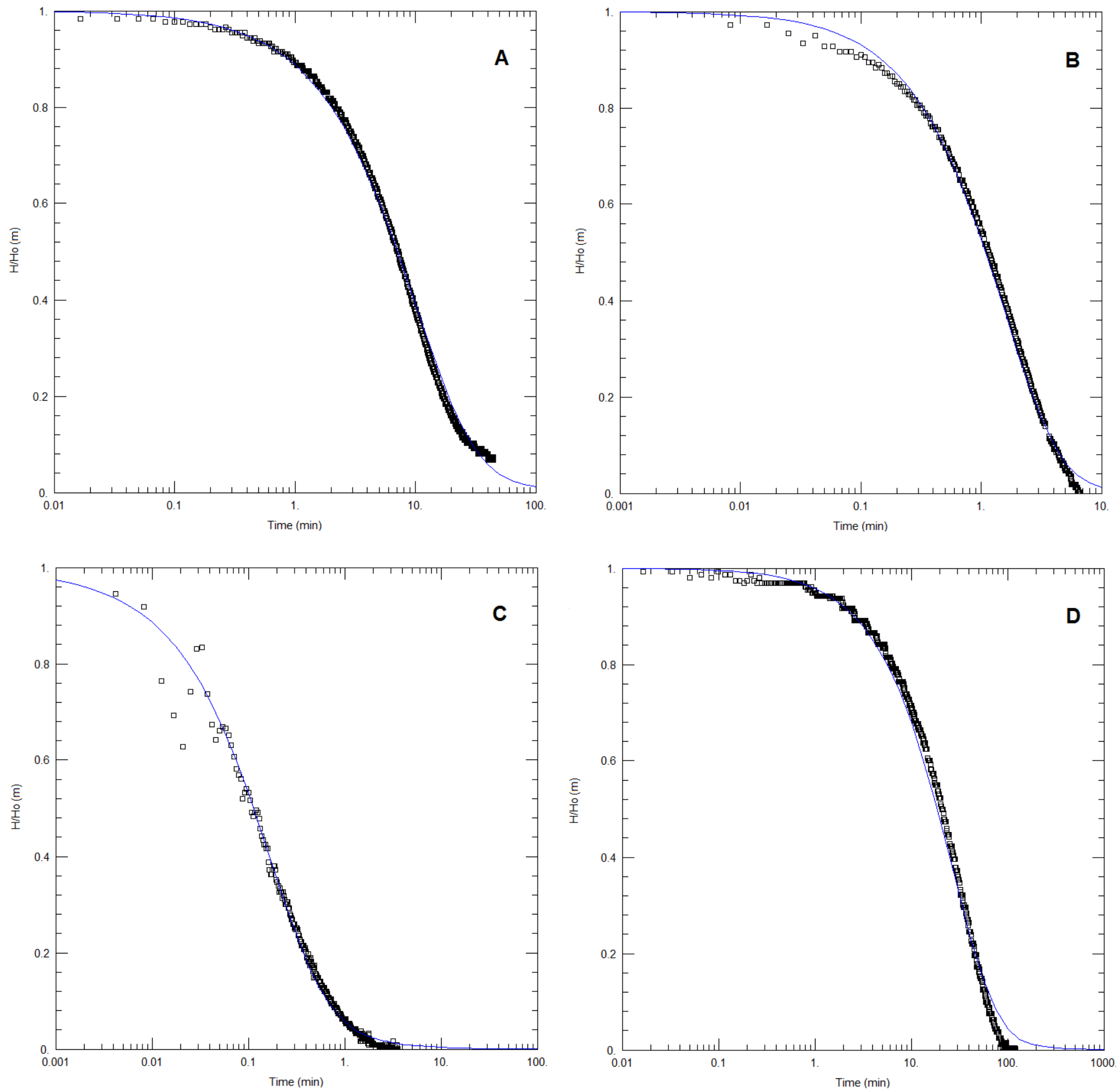


Figure 5. (a) Ideal representation of a well during a slug in test. (b) Cooper et al. (1967) family of type curves. Modified from Cooper et al. (1967).



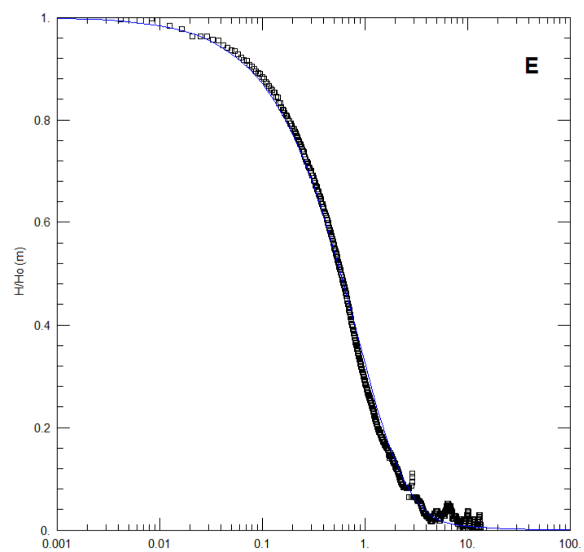
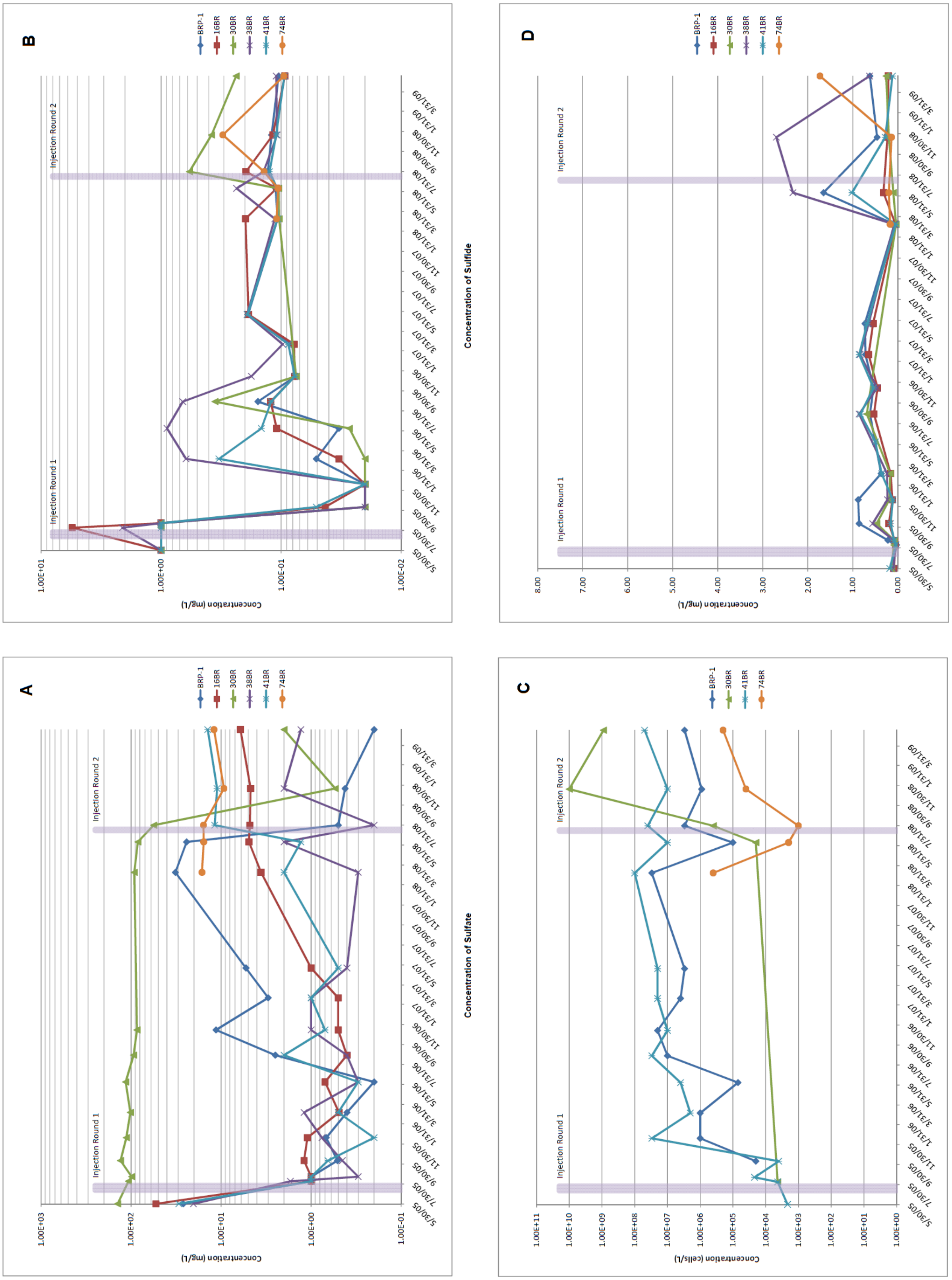


Figure 6. Normalized water-level displacement (H/H_o) over time, fit to Cooper et al. (1967) type curves (blue lines). (a) 16BR. (b) 17BR. (c) 41BR. (d) 74BR. (e) BRP-1.

Figure 7.(next page) Concentrations of (a) sulfate, (b) sulfide, (c) *Dehalococcoides* (DHC), and (d) dissolved oxygen, at select biostimulation wells. Modified from deFlaun, unpublished.


A
Concentration of Sulfate
Concentration (mg/L)
Injection Round 1
Injection Round 2
BRP-1
16BR
30BR
38BR
41BR
74BR
B
Concentration of Sulfide
Concentration (mg/L)
Injection Round 1
Injection Round 2
BRP-1
16BR
30BR
38BR
41BR
74BR
C
Concentration of DHC
Concentration (cells/L)
Injection Round 1
Injection Round 2
BRP-1
30BR
41BR
74BR
D
Concentration of Dissolved Oxygen
Concentration (mg/L)
Injection Round 1
Injection Round 2
BRP-1
16BR
30BR
38BR
41BR
74BR

Discussion

The most significant T decrease occurred at well 30BR. In 1997, T was approximately 0.6 m^2/day at 30BR, while T was too low for measurement in 2012 (table 1). Assuming T was <0.007 m^2/day, T in 2012 must have decreased by at least two orders of magnitude at 30BR. This is a considerably larger decrease than 16BR, 17BR, and 41BR, which decreased within one order of magnitude (table 1). Slug test analysis is quite subjective (Butler, 1998), so any T change within one order of magnitude may result from this subjectivity and not represent a true change of T (Daniel J. Goode, personal communication, 2013). Since the T decreases at 16BR, 17BR, and 41BR are each within one order of magnitude (table 1), the changes are likely minor and negligible. The decrease of T at 41BR is slightly greater than 16BR and 17BR, likely due to the relatively poor fit of early time data to the type curve (fig. 6) resulting from a relatively less "clean" slug removal during testing. The two order of magnitude decrease at 30BR probably signifies the only true change of T, and indicates biofouling-induced clogging is most likely to have occurred in 30BR.

Sulfate-reducing bacteria (SRB) are frequently identified as culprits of biofouling in anoxic groundwaters. Cullimore (1999) describes growth of biofilms and precipitation of metal sulfides as materials associated with SRB biofouling. A frequently formed metal sulfide is ferrous iron sulfide ("FeS"), which precipitates as SRB reduce sulfate to sulfide, then subsequently reacts with ferrous iron dissolved in groundwater (Cullimore, 1999; Kennedy et al., 2006 ; Gramp et al, 2010), a relatively quick reaction (Rickard, 1995; Hansson et al 2006; Kennedy et al., 2006). The resulting FeS is typically in the form of amorphous FeS or poorly crystalline mackinawite (Anderko and Shuler, 1997; Hyun and Hayes, 2009; Gramp et al., 2010). Groundwater at NAWC is anoxic, and sulfate-reducing conditions are dominant (Chapelle et al., 2012), so if biofouling has occurred in 30BR, it likely developed due to SRB activity and the subsequent build-up of SRB biomass and/or FeS in the fractures intersecting the well screen. Biomass clogging frequently occurs at injection wells utilized during bioremediation (MacDonald et al., 1999; Lutes et al., 2003). 30BR was as an injection well during the 2008 biostimulation, which renders the fractures more vulnerable to clogging by biomass. FeS has also been identified in clogging materials in groundwater wells (Barnes and Clarke, 1969; Walter, 1997; Houben, 2003; Hoelen et al., 2006; Smith and Roychoudhury, 2013), and is therefore a possible clogging material in 30BR as well. However, the possible presence of SRB biomass and FeS in these cases does not indicate whether their formations caused any change in hydraulic properties, as their presence may have no effect. For example, an injection experiment by Kennedy et al. (2006) produced FeS from SRB biostimulation, but observed no change in matrix permeability as preexistent iron minerals were converted to FeS.

The T estimate at 30BR in 1997 is less than those at all other biostimulation wells in 1997 except for 38BR (table 1). NAWC wells with low T will contain fewer and/or smaller intersecting fractures (Lacombe and Burton, 2010; Tiedeman et al., 2010). Possible clogging from enhanced SRB activity most likely occurred in smaller fractures where a mass of biofouling material can have a more significant impact on overall hydraulic properties. A smaller conduit is more likely to be clogged than a larger conduit, so the T decrease will be more apparent. Thus, well 30BR is surely relatively less fractured and more susceptible to clogging than 16BR, 17BR, 41BR, and BRP-1. Well 38BR is likely very susceptible as well, but since T could not be measured, the extent of possible biofouling would be unknown.

Groundwater chemistry data routinely collected by Geosyntec Consultants™ and USGS in the biostimulation wells (deFlaun et al., 2006; deFlaun, unpublished) further indicates SRB biofouling in 30BR. Well 30BR was not included in the biostimulation until the 2008 injection. After the 2008 injection, 30BR concentrations of sulfate decreased (fig. 7a) and concentrations of sulfide (fig. 7b) and *Dehalococcoides* bacteria (DHC) increased (fig. 7c) while dissolved oxygen (DO) remained low (fig. 7d). The change of sulfate to sulfide in low oxygen groundwaters is a common sign of SRB activity. The increase of DHC indicates EVO injection was successful in attracting bacteria to the well. EVO acts as an electron donor for both DHC and SRB, so if DHC increased, SRB counts may have grown as well, explaining the increased sulfate to sulfide transformations. As previously stated, sulfate-reducing conditions are widespread at NAWC (Chapelle et al., 2012), so the increased abundance of SRB in the biostimulation wells is realistic. However, an increase in SRB count does not necessarily indicate biofouling, as their presence may be unrelated to the T decrease.

Likewise, an observed decline in hydraulic properties may result from the build-up of other materials even if SRB materials are present, or from a different process entirely. A number of additional factors may contribute to, or solely generate, the observed decrease of T in 30BR.

DHC biomass can accumulate alongside SRB biomass and further clog the fractures. Since the EVO injection travels with groundwater flow through the fractures, the clogging may arise from EVO itself; 30BR is less fractured, so the well is more likely to possess poor hydraulic connections to recovery wells. A large concentration of EVO may have lingered in 30BR fractures without escape and caused a blockage in the fractures. The T decrease can also be unrelated to biostimulation; fracture collapse and clay infilling, for example, can also trigger a decrease of T.

Conclusion

Slug tests of wells utilized during the 2005/2008 biostimulation at NAWC were successful in observing substantial decreases of T between 1997 and 2012. Wells 16BR, 17BR, 41BR, and BRP-1 each displayed absent or negligible changes of T that likely stem from the subjectivity of slug test analysis and operating procedure. T at well 30BR decreased at least two orders of magnitude after the injection, which is likely a true decrease of T. Groundwater chemistry data indicates sulfate-reducing bacteria may be responsible for the biofouling of well 30BR by accumulation of biomass and/or precipitated FeS within fractures. Fractures intersecting 30BR are more susceptible to clogging than those of 16BR, 17BR, 41BR, and BRP-1, because the fractures are smaller, fewer in number, and likely provide relatively poor connections to other parts of the aquifer.

While slug tests are effective in observing changes in hydraulic properties, attributing the changes to biofouling or any other cause is only conjecture. Causal explanations cannot be provided without implementation of additional tests. For example, logging well 30BR with a high-definition optical televiewer and borehole video camera can provide visual images of the well and help visually distinguish possible clogging materials. Acquiring physical samples of these materials for laboratory analysis is necessary to verify their identity. These confirmations will advance understanding of how biostimulation can modify hydraulic properties around a well in fractured rock aquifers.

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