Water Quality Standards

Turbidity Technical Review

Summary of Sources, Effects, and Issues Related to Revising the Statewide Water Quality Standard for Turbidity

State of Oregon
Department of Environmental Quality

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Executive Summary

The Oregon Department of Environmental Quality (DEQ) is revising its water quality standard for turbidity, which has been largely unchanged since 1977 and currently prohibits more than a ten percent cumulative increase in natural stream turbidities relative to an upstream control point. This document is a summary of the science regarding turbidity and its effects, which served the development of policy choices prepared by DEQ and discussed with an advisory workgroup of parties interested in the development of the standard. Policy choices are discussed in a separate document.

Turbidity is a relative measurement of reduced visual clarity (scattering and absorption of light by particles in water) as compared to a calibrated standard. Increased turbidity levels are caused by suspended particles, dissolved organic matter, and planktonic organisms in the water column. There are many different types of nephelometric instruments which measure turbidity. They differ by the type of light used (white light vs. infrared), whether they have one or many light detectors, whether they measure backscatter, and by use of one or multiple beams of light. Different turbidimeters, even those employing the same type of instrumentation, often report different measurements even on the same sample. These differences are small or undetectable at low levels of turbidity (less than approximately 10 NTU), but are larger at higher levels of turbidity. Other measures of water clarity exist including black disk measurement, light transmissometry and Secchi Depth. Turbidity is generally highly correlated with total suspended solids readings; a number of studies have established relationships between the two for specific Oregon waters.

Natural weathering and decomposition of rocks, soils, and dead plant materials and the transport or dissolution of the weathered products in water contributes a natural “background” of turbidity-causing suspended and dissolved materials to natural waters. Large fluctuations of turbidity can be caused by natural disturbances or episodic events, such as fires, floods, and landslides. Throughout Oregon, background turbidity during the summer is quite low (1 NTU in most ecoregions; 2 NTU in the Willamette Valley ecoregion). The exception to this pattern is in Hood River and potentially other streams dominated by glacial melt. Upper reaches of the Hood River experience a diurnal pattern of turbidity with peak turbidities occurring during peak melt late in the afternoon. Turbidities often reach in the hundreds during this time. In lower reaches of Hood River where glacial melt is mixed with typical snowmelt or rainwater systems, summer turbidity is still higher than other areas of Oregon with a median of approximately 7-8 NTU based on available DEQ data.

During the wet season, median turbidity in most undisturbed watersheds in Oregon is below 5 NTU, although there are exceptions, such as higher turbidities in the Williamson River during the spring. In disturbed watersheds, such as the Tualatin basin or Johnson Creek, turbidity is either “flashy” or more persistently high. In general, during rain events in Oregon, turbidity in small, forested streams increases as stream discharge increases and peaks slightly earlier than discharge peaks; this pattern is called the “hysteresis effect.” Departures from this effect are often correlated with disturbances in the watershed.

For this report, DEQ conducted a “concentration-duration-frequency” analysis on USGS turbidity data from three streams. Such an analysis identifies the number of times in a given period (three years for our analysis) turbidity exceeds a certain threshold continuously for a certain amount of time. The analysis indicated that natural levels of turbidity occasionally exceed levels that are shown to have adverse effects. In examining the effects of turbidity on beneficial uses, DEQ generally relied on studies that were conducted within Oregon or the Pacific Northwest. DEQ also utilized studies examining the effect of turbidity on macroinvertebrates, primary productivity, recreational “usability,” and aesthetics in New

Zealand, as we concluded that the studies done there were sufficiently robust for consideration and that water quality characteristics in New Zealand are comparable to those in Oregon.

Elevated turbidity in streams and lakes has been shown to affect primary productivity (growth of algae and submerged macrophytes) in streams, lakes, and estuaries. However, increased photosynthetic efficience can temporarily counteract this effect, although potentially at a cost to growth. Research conducted in New Zealand indicates that algal production is decreased at turbidity levels of 8 NTU as compared to clear (1 NTU conditions). Turbidity also has been found to limit growth of macrophytes in lakes; however, there is insufficient data to determine a specific turbidity level that would correspond to decreased growth. In Oregon estuaries, research has examined how turbidity may affect growth of eelgrass (*Zostera marina*). EPA has recommended water clarity criteria to protect eelgrass growth in the Yaquina Bay Estuary, but not in other estuaries due to additional data needs and the influence of other variables, such as salinity.

Increased turbidity is correlated with various metrics of decreased benthic macroinvertebrate abundance and diversity, as well as populations of zooplankton. There are two ways in which turbidity may affect such populations: 1) turbidity may reduce food availability for primary consumers by limiting primary production and 2) increased turbidity and suspended sediment may increase drift of macroinvertebrates due to clogging of benthic habitat. Studies in Oregon indicate that macroinvertebrate abundance and diversity are affected at turbidities of approximately 4-8 NTU as compared to reference conditions (1-2 NTU). Studies at the lower end of this range focus on abundance and diversity of Ephemeroptera, Plecoptera, and trichoptera (EPT) species. However, DEQ data indicate that there is fairly weak correlation between the presence and abundance of these species and indices of biological integrity for fish. As a result, there is some uncertainty in this range (4-8 NTU) as far as the extent to which they affect aquatic life.

Turbidity decreases reactive distance, the distance at which fish detect and orient themselves toward prey. Studies indicate that this effect, in turn, results in decreased feeding success in salmonids in short trials and in decreased growth after exposures to moderate turbidities (20 NTU) after two or three weeks. There is uncertainty as to what the minimum effect level is for decreased growth in salmonids, as even the lowest turbidities tested in studies resulted in significant effects. Studies indicate that salmonids exposed to moderately high turbidity levels in natural settings are able to feed in the benthos, although possibly at a lower rate and with increased energy expenditure due to a more active foraging strategy.

Some studies indicate that fish populations are impaired (decreased density, smaller, or lack of sediment intolerant fish) in areas with chronic turbidity; however these studies lack sufficient data that could be useful for setting a water quality standard for turbidity.

A few studies have linked increased turbidity with other behavioral effects in fish, such as changes in territorial behavior, avoidance of turbid water, and increases in blood sugar levels; however, in some cases, it may be difficult to separate the visual effects from direct effects of suspended sediment.

Several studies have documented the use of turbid waters by juvenile fish as cover from prey. Some of these studies also have shown that streamside vegetation appears more important than “cloudiness” as cover. Moreover, models indicate that the use of “cloudy” water is more than offset by the loss of feeding efficiency, unless accompanied by an increase in food availability.

There is a limited body of research on the effects of reduced water clarity on the desire of people to recreate in streams and lakes. Studies on turbidity effects on aesthetics and swimming are primarily limited to surveys conducted in New Zealand. These surveys show that relatively low (~3 nephelometric turbidity units, NTU) turbidity levels are considered unsuitable for swimming and aesthetic purposes.

As turbidity in drinking water source areas increases, the cost to meet Safe Drinking Water Act-mandated turbidity levels similarly increases due to increased material and maintenance costs. In addition, some public water systems in Oregon using filtration systems must shut down their operations when source waters exceed 5 NTU.

In summary, the literature indicates that chronic and low levels of turbidity (as low as 4 NTU) are correlated with adverse effects on aquatic life, such as reduced invertebrates. Such effects may cascade into higher trophic levels, resulting in population-level reductions to fish. Reactive distance of fish decreases with increasing turbidity levels; consequential effects on fish growth and feeding generally are reported around 20-25 NTU for exposures lasting two or three weeks. However, studies have not tested effects at turbidity levels lower than this. At the same time, studies have documented fish feeding even at relatively high turbidities. Studies indicate that turbidity as low as 2-5 NTU can affect people's perception of the desirability of waters for recreation. Increased suspended sediment levels that are associated with turbidity have a small effect on drinking water treatment costs; however, levels as low as 5 NTU can cause some drinking water treatment operators to shut down their operations. If this occurs frequently enough, municipalities may have difficulty providing safe drinking water to their residents.

Chapter 1. Introduction

This Oregon Department of Environmetnal Quality is reviewing scientific literature and data to support its process to revise the water quality standard for turbidity (Oregon Administrative Rule 340-041-0036). DEQ is revising the standard to incorporate the best available science regarding the effects of turbidity on beneficial uses of Oregon waters, account for natural variability in turbidity, and create a standard that can be implemented across Oregon's Clean Water Act programs. The purpose of this document is to summarize data and literature that are relevant to the effects of turbidity on beneficial uses of Oregon waters and describe natural variabiilty.

The water quality standard for turbidity was last reviewed 2003-2006. Although a proposed water quality standard for turbidity underwent public notice and hearings, DEQ ultimately did not propose a revised rule to the Environmental Quality Commission. This document builds on the research and information in DEQ's 2005 *Draft Technical Basis for Revising Turbidity Criteria*. This document incorporates additional scientific literature published since 2005, additional literature that was not considered in the earlier document, and data and literature that DEQ received as a result of a Call for Data sent out to interested parties in early 2010. In addition, this document addresses, to the extent data and information are available, comments DEQ received from the Independent Multidisciplinary Science Team, a scientific panel that advises the State of Oregon on matters of science related to fish recovery, watershed health, and water quality improvements.

Despite a fairly strong body of literature, there is still considerable uncertainty with respect to the effects of turbidity on beneficial uses, particularly when looking at effects on aquatic life. In addition to presenting the literature and some supporting analysis, DEQ has included a discussion of data gaps and uncertainty with respect to various categories of effects. Such information will inform policy choices regarding the appropriate criteria to protect beneficial uses. These choices were discussed with a stakeholder advisory group as rule language was developed.

DEQ's review focuses on the direct effects of turbidity (reduced light penetration) on beneficial uses; it does not specifically address direct effects of suspended sediment (e.g., fish mortality), bedded sediment (e.g., egg survival), nor other water quality parameters often associated with turbidity, such as toxics, nutrients, or bacteria for the following reasons:

- Levels of suspended sediment that result in direct mortality are generally very high (thousands of turbidity units); DEQ's revised standard will focus on sublethal effects and thus also protect from lethal effects.
- Bedded sediment, which may in some cases be correlated with turbidity, is covered by DEQ' s narrative sediment standard at OAR 340-041-0007(12). DEQ conducted an effort to interpret this standard numerically in 2009; while this effort was not finalized, DEQ would at some point like to finish this effort.
- DEQ already has water quality standards for bacteria and toxics. While DEQ does not have nutrient standards, the agency generally develops nutrient targets in areas where nutrients are contributing to impairments of other water quality standards, such as dissolved oxygen, pH, and chlorophyll *a*.

In some cases, it is difficult to separate out the effects of turbidity from direct effects of suspended sediment, which is generally highly correlated with increased turbidity levels. In particular, associations of increased turbidity with decreased macroinvertebrate populations may result from two different mechanisms: 1) decreased availability of plankton due to reduced light in the water column, which is, directly related to turbidity; and 2) increased drift of macroinvertebrates due to sediment suspended in the water column. However, it is difficult, if not impossible, to separate out these two mechanisms in research. Therefore, we have included this endpoint in our discussion of effects of turbidity on aquatic life.

Another effect that, while not directly related to reduced light penetration, is considered in this report is the effect of suspended particles on treatment of domestic water supplies. Such effects are generally reported in the literature in terms of turbidity and controlled under the Safe Drinking Water Act through limits placed on turbidity in finished drinking water.

DEQ used this document to inform the development of water quality standard regulations based on best available science. This rule development effort will ultimately require some difficult policy choices, which will be discussed by a stakeholder advisory group as the rulemaking moves forward.

Scope of the Turbidity Water Quality Standard Review

Chapter 2 of this paper focuses on the basic properties of turbidity, including its definition, measurement, and natural variability. In addition, Chapter 2 presents some analysis of Oregon turbidity data as a starting point for discussing baseline conditions.

The remainder of the review focuses on effects of increased turbidity and reduced light penetration on the following endpoints:

- Aquatic life (Chapter 3)
  - Primary productivity in the light column
  - Invertebrate density and diversity
  - Effects on fish prey-predator dynamics
  - Effects on fish growth
  - Non-feeding effects on fish
  - Fish population dynamics
- Recreation (swimming and aesthetics) (Chapter 4)
- Domestic water supply (Chapter 5)

In the discussion of aquatic life effects of turbidity, the review separates the discussion into effects on aquatic life in streams, lakes/reservoirs, and estuaries. This information will assist DEQ in determining whether separate water quality criteria for these types of waters are appropriate for different water body types.

Geographic Scope of Literature Search

In examining effects of increased turbidity levels on reduced primary productivity in streams and lakes, DEQ examined studies that were conducted worldwide, in part because the literature has noted that responses of aquatic plants (phytoplankton and macrophytes) to reduced light penetration is fairly consistent; moreover, only one Oregon data set was identified. In examining effects of turbidity on primary productivity, particularly growth of submerged aquatic vegetation (SAV) in estuaries, DEQ's review focused on studies conducted in the Northwest, as research has shown that differences in tidal ranges and regimes, temperature, and winter irradiance make it inaccurate to extrapolate studies from other locations.

DEQ's discussion of the effects of increased turbidity levels on fish feeding dynamics and other behavioral affects generally focuses on research done on fish that are found in Oregon, and in particular, salmonids. Effects of turbidity on feeding, in particular those related to the ability to detect prey, can depend on the sensory mechanisms and capabilities of predator and prey (Ljunggren and Sandström 2007; B.C. Harvey *pers. comm.*). As a result, studies focusing on fish that are not present in Oregon, are only useful for illustrative purposes, but not for setting a water quality standard.

Effects of reduced visual clarity on the desire of people to use waters to recreate is limited to consideration of a handful of studies conducted in New Zealand, as this was the only relevant research DEQ was able to identify. Such studies merit consideration, as local conditions in New Zealand and Oregon (in particular, west of the Cascades) are sufficiently similar, indicating that their perception of what can be considered "good" water for swimming and aesthetics should be similar (R. Petersen, *pers. comm.)*. DEQ has also included a brief discussion of effects of increased turbidity levels on decreased catch rates for fishing, although information available on this topic is limited to anecdotal reports.

DEQ's review of the effects of increased turbidity levels on drinking water focused on a handful of economic studies in the U.S., including one in the Willamette Valley, that examine the relationship between increased turbidity and drinking water treatment costs. DEQ also reviewed the findings of a recent study it conducted that examined changes in turbidity patterns at eight public water systems in Oregon's North and Middle Coast Range. The study examined how changes in turbidity levels can affect these systems and how human impacts can minimize or worsen those changes (DEQ 2010).

DEQ recognizes that turbidity correlates well with other pollutants and is therefore sometimes considered a good surrogate for those pollutants in determining water quality (Figure 1). For example, suspended sediments that increase turbidity levels can also be important transporters of nutrients, bacteria and toxic compounds (Sorensen, et al. 1977). While cognizant of these relationships, DEQ considers that these parameters are best addressed as separate narrative or numeric criteria (e.g., toxics; excess fine bedded sediment; inter-gravel dissolved oxygen).

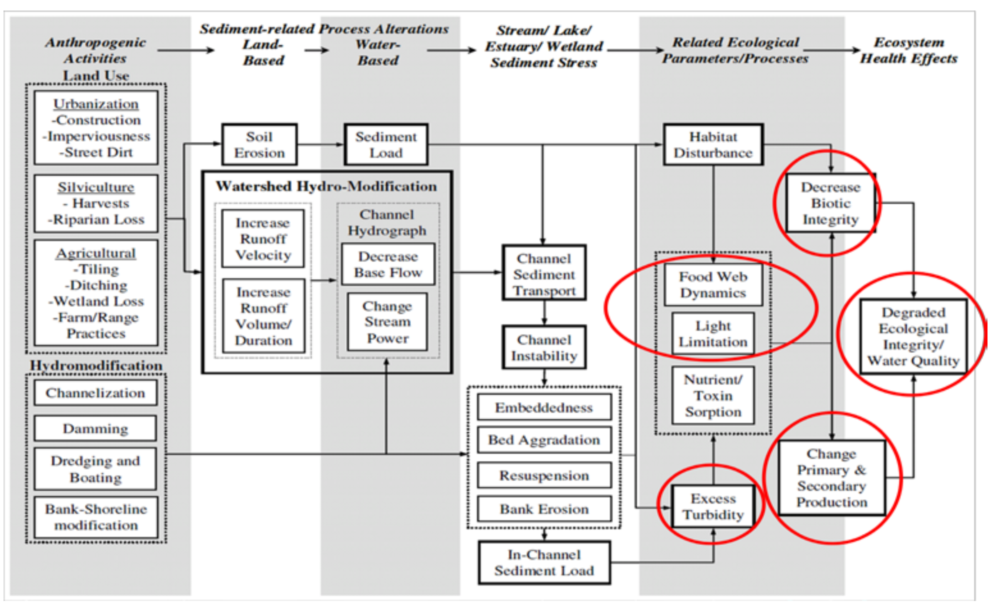

Figure 1. Focus of aquatic life effects considered in this review (from US EPA, 2006)

Chapter 2. Overview of Turbidity

Definition of turbidity

Turbidity measures the “cloudiness” of water; more precisely, it measures the extent to which light is scattered and absorbed by suspended sediment, dissolved organic matter, and, to a lesser extent, plankton and other microscopic organisms (Clesceri, et al. 1994). From a technical standpoint, turbidity is a relative measurement of scattering as compared to a calibrated standard, usually a formazin suspension (Davies-Colley and Smith 2001). Turbidity is also referred to as the inverse of the “clarity” of water. Light that is not scattered or absorbed by turbidity-causing particles passes through the water. In other words, increased turbidity reduces the distance that light can penetrate into the water column.

Measurement of turbidity and other expressions of clarity

This section describes different types of turbidimeters, as well as other methods for measuring water clarity. Beginning in the early 20th century, turbidity was measured using a Jackson candle turbidimeter, which consisted of a special candle and a flat-bottomed glass tube (Sadar 1996). The Jackson turbidimeter was calibrated by a series of standard suspensions of known clarity using diatomaceous earth in distilled water. Measurement was made by slowly pouring a turbid sample into the tube until the image of the candle flam diffused to a uniform glow (Sadar 1996).

Jackson turbidimeters cannot measure turbidity lower than 25 JTU, are cumbersome, and depend on human judgment to determine the extinction point. Eventually, nephelometric detectors were developed and became the accepted method to measure turbidity. Nephelometric devices measure light scattering through a restricted range of angles to the incident light beam relative to a standard suspension, usually of formazin. Several different nephelometric methods involving different light sources and detector arrangements have been developed to measure turbidity. Some instruments are “ratiometric,” with multiple detectors arranged at various angles. These instruments then calculate turbidity using a ratio of the light received by the different detectors. The United States Geological Survey (USGS) has developed a data reporting protocol based on the type of light source and detector arrangements of various turbidity instruments (Anderson 2005).[1] The headings for each of the instruments described below include reporting units using the USGS protocol.

Non-ratiometric, white light (Nephelometric Turbidity Units, NTUs)

The most common type of nephelometric instrument reported in the literature is a white light turbidimeter, which has a single detector centered at 90° from the incident light path. Non-ratiometric, white light turbidimeters are compliant with EPA Method 180.1 for determining turbidity by nephelometry, which requires that the light source for the nephelometer be a tungsten lamp (white light) operated at a color temperature of 2200-3000° K and that the detector is centered at 90° from the incident light path and does not exceed ± 30° from 90° (EPA 1993).[2] The accepted range of measurement for such meters is 0-40 NTU.

[1] The protocol is primarily a way for the USGS to report and track their own turbidity data (Chauncey Anderson, *pers. comm.*)
[2] NTRUs also fit within the definition of an appropriate turbidimeter under 180.1.

Ratiometric, white light (Nephelometric Ratiometric Turbidity Units, NTRUs)

The ratiometric, white light turbidimeter design is also considered compliant with EPA Method 180.1. The difference between a ratiometric and non-ratiometric instrument is additional photodetectors located at angles other than 90° from the incident light. The ratiometric turbidimeter combines signals from each of these detectors mathematically to calculate the turbidity of the sample. Ratio nephelometers purportedly perform better with colored samples than traditional nephelometers (EPA 1999).

Near infra-red (IR) light, non-ratiometric and ratiometric (Formazin Nephelometric Unit, FNU and Formazin Ratiometric Turbidity Unit, FNRU

Near IR instruments utilize a light-emitting diode with wavelength 860 ± 60 nm. The detector angle must not exceed ± 2.5° from the 90° incident path. These types of nephelometric turbidimeters are compliant with the ISO 7027 standard, which is commonly used in Europe, but not EPA method 180.1, due to the type of light. Most USGS continuous turbidity monitoring stations in Oregon use this methodology.

Backscatter/ratiometric turbidimeters, white light or near IR light (Backscatter Unit, BU or Formazin Backscatter Unit, FBU)

Backscattter turbidimeters use incident beams at 30°±15° to the incident sample for high levels of turbidity and nephelometric detection (90° angle) for low-levels. Such devices determine turbidity using light scatter from or near the surface of a sample. These types of meters are most appropriate for high-level turbidities (up to 10,000 units).

Multiple-beam turbidimeters, white light or near IR light (Nephelometric Turbidity Multibeam Unit, NTMU or Formazin Nephelometric Multibeam Unit (FNMU)

Multiple-beam turbidimeters have multiple light sources and detectors to provide reference and active signals with at least four independent measurements for each reading. The final reading is determined with a ratio algorithm.

Other methods for measuring cloudiness or visual clarity

Light transmissometry

In contrast to turbidity, transmissometry measures light extinction in a water column as a function of both scattering and absorbance of light from a sealed submersible light source and a detector optimized for maximum transmission *in situ* by a selective filter. Transmissometers display data as percent transmission or as a volume attenuation coefficient (Telesnicki and Goldberg 1995). Some authors have expressed a preference for using transmissometry over nephelometry for measuring visual clarity because it is an absolute measurement and can be used to calculate a scattering coefficient which is more explicitly related to suspended solids concentrations (Davies-Colley and Smith 2001). In Oregon, research examining the effect of light and shading on the growth of submerged aquatic vegetation (SAV) often utilizes both turbidity and light transmissometry data (e.g., Brown, et al. 2007). However, while light transmissometry has been used for research, they have not been used widely for regulatory purposes.

Secchi Depth

Since the 19th century, water clarity has been measured in lakes, reservoirs, and estuaries using a Secchi Disk. The Secchi Disk is an alternating black-and-white disk with a 30 cm diameter that is lowered into water by a rope until the disk is judged to disappear from view. Secchi depth, z_{SD} provides a simple (and inexpensive) indicator for the clarity of natural waters (Preisendorfer 1986). Secchi depth can vary depending on the reflectance of the white face of the disk and the reflectance of the water. Secchi depth readings are thus dependent on lighting conditions and are difficult in shallow systems (Davies-Colley and Smith 2001). Smith (2001) has recommended procedures for increasing precision.

Black disk measurement
Recently, researchers, particularly in New Zealand, have utilized black disk measurement to measure turbidity. Black disk measurement is the maximum sighting distance of a perfectly black target, viewed horizontally, instead of the vertical measurement of the Secchi disk. Because the target is viewed horizontally, black disk measurement can be used in both shallow and deep waters. Researchers have used black disk measurement to estimate a beam attenuation coefficient with reasonable precision at a wide range of conditions (Davies-Colley and Smith 2001). The relationship between turbidity and black disk measurements is still site-specific. This method has not gained wide use in Oregon, if at all.

Relationship of suspended solids with turbidity
While total suspended solids does not measure clarity, it is generally well correlated with turbidity. Turbidity is often a less expensive alternative to measuring suspended solids (Gippel 1995). However, the relationship between turbidity and solids is confounded by variations in particle size, particle composition, and water color (Gippel 1995). As a result, there is no universal relationship between turbidity and suspended solids. In Oregon, site-specific relationships have been developed for a number of sites including the Santiam River Basin (Uhrich and Bragg 2003) and Oak Creek and Flynn Creek watersheds (Beschta 1980). However, even these relationships can vary from storm-to-storm, seasonally, and from year-to-year (Beschta 1980).

Variability among turbidimeters and turbidity units
Turbidimeters, even those employing the same light source and detector arrangement, can produce different turbidity readings for the same water sample due to optical differences, calibration techniques, instrument design, and the user. As a result, there is some uncertainty to effects levels expressed in the literature, especially as some of the papers cited do not report which type of meter was utilized to measure turbidity. However, literature indicates that the difference between instruments becomes more pronounced at higher turbidity levels; below approximately 10 NTU, different instruments are fairly consistent in turbidity measurements (Telesnicki and Goldberg 1985). Moreover, studies in which turbidity is measured multiple times over a longer time period may reduce uncertainty as compared to studies in which a single sample is taken.

Summary of literature
In an early study, Duchrow and Evenhart (1971) found that the relationship between turbidity and concentration of solutions of seven different materials differed among three different types of turbidimeters. In a study comparing turbidimeters of different technologies, Gippel, et al. (1991) found that an attenuance turbidimeter gave absolute readings of 2.5 to 4 times higher than a nephelometric turbidimeter despite being identically calibrated. The authors noted that the attenuance turbidimeter was more sensitive to the presence of color, particularly at low levels of suspended sediment.

Davies-Colley and Smith (2001) compared turbidity readings from a non-ratiometric and a ratiometric nephelometer from the same manufacturer, on 77 water samples from New Zealand rivers. The results from the study indicate that readings from the ratio nephelometer were consistently higher by about 30%. Barter and Deas (2003) tested readings of primary formazin standards (five replicates for each of six standards) by five portable and found that coefficients of variation between the meters ranged from 1.5 to 6.8%. Differences were within 13% for NTU calibrations of 400 NTU or under, but 21% for 800 NTU. When using the instruments to test the turbidity of various effluents and receiving waters, coefficients of variations ranged from 6.6% to 44.1%. Higher coefficients of variation were associated with the samples with the low mean turbidity readings (0.14-1.6 NTU) and high or very high mean turbidities (66.9-506.04 NTU).

Lewis, et al. (2007) studied measurements from eight turbidimeters (two IR-light, backscatter instruments; four IR-light, nephelometric instruments; and two white light, nephelometric instruments) of samples created with filtered sediment from 10 Coast Range watersheds in California. The study found that the mean error between sensor pairings was 12%, but maximum errors occasionally exceeded 100%. The authors concluded that sensors that conform to the same standards do not necessarily give similar turbidity readings; that relationships between sensors of the same design were more consistent, for different sediments, than relationships between sensors that used different methods on the same sample; and that conversion of *in situ* sensor readings to laboratory readings is prone to relatively large errors unless the laboratory meter is set to use the same method as the *in situ* meter.

Both U.S. EPA method 180.1 and ASTM standard D7315-07 recommend diluting samples exceeding 40 turbidity units and adjusting the turbidity measurement proportionally to the dilution (e.g., for a 5:1 dilution measuring of 30 NTU, turbidity would be reported as 180 NTU) (USEPA 1993; ASTM 2007). However, dilutions may change the matrix of the sample, which may skew turbidity values of the sample (M. Sadar, HACH Corporation, *pers. comm.*) and result in an incorrect value (which would then be compounded when multiplying to reflect the original concentration). This may be somewhat counteracted by using filtered water from the sample site to dilute the sample.

Human error may also play a big part in variation of turbidity measurements. Landers (2002) asked fourteen participants in a sediment workshop to calibrate nine turbidimeters and measure samples of three different concentration/substrate combinations. The results indicated fairly high coefficients of variation ranging from 21% for samples with a sediment concentration of 150 mg/L (median turbidity 53 NTU) to 42% for a samples with sediment concentrations of 600 mg/L (equivalent to 268 NTU) and 93-94% fines. The study indicated that factors associated with the operator, sub-sampling, and other factors in an uncontrolled environment could contribute to variability.

Variability in Turbidity

Natural weathering and decomposition of rocks, soil, and dead plant material, and the transport or dissolution of the weathered products in water, contribute a natural "background" of turbidity-causing suspended and dissolved materials to natural waters (Sorensen, et al. 1977). Large fluctuations of turbidity can be caused by natural disturbances or episodic events, such as fires, floods, and landslides.

Turbidity increases with stream discharge, generally corresponding to storm events that carry more sediment; however, relationshiops between suspended sediment and turbidity can change from storm-to-storm, seasonally, and from year-to-year (Beschta 1980). The first storm following the summer dry period generally results in higher turbidity than subsequent larger flows due to an initial flush of suspended-sediment (Paustian and Beschta, 1979). In forested watersheds, peak turbidity occus before peak discharge. This effect is termed "hysteresis" (Bogen 1980).

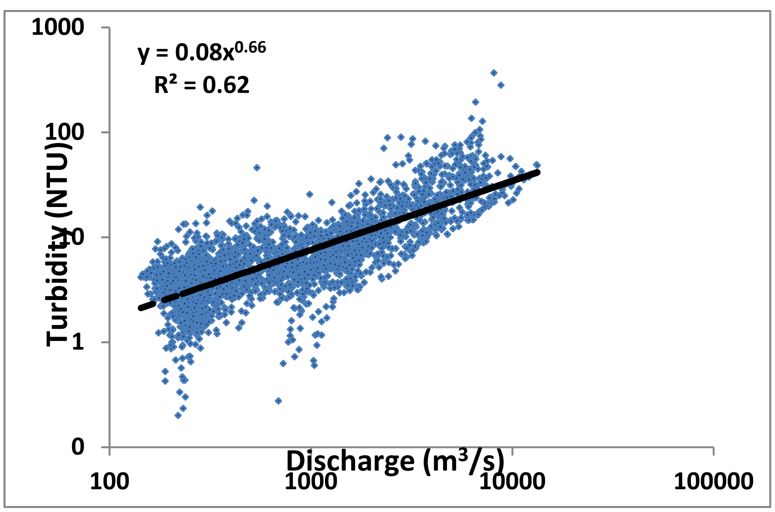

Figure 2. Turbidity vs. Discharge graph for Tualatin River, 2004-2014. (R. Beschta, pers. comm.)

As discharge subsides, turbidity returns to background levels. Figure 2 shows the relationship between discharge and turbidity in the Tualatin River between 2004 and 2014. The figure highlights both the general pattern of correlation, but also the variability in turbidity for any given discharge. For example, at a discharge of approximately 1000 m^3/s, turbidity ranges from 1 NTU to 25.6 NTU.

Elevated turbidity may persist after a storm as compared to pre-storm turbidity. For example, following a storm event in late January/early February 2000, turbidity at Blowout Creek in the North Santiam River continued to be elevated, while turbidity at other sites in the watershed fell to lower levels. The persistence of elevated turbidity in this case is likely due to the presence of a clay-rich natural debris flow (Uhrich and Bragg 2003; Figure 3). Smaller clay-sized particles remain suspended for a longer time than larger particles, particularly in a flowing stream.

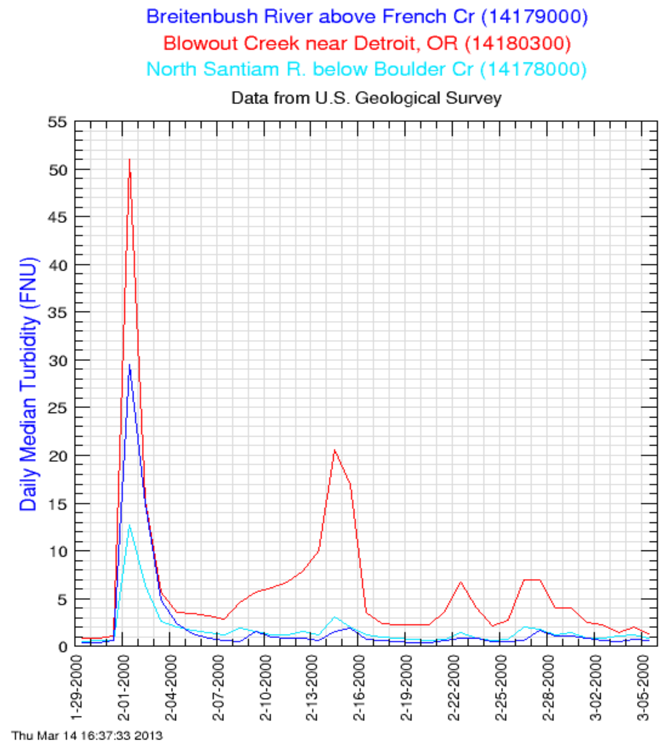

Figure 3. Daily median turbidity during and after a storm event at 3 North Santiam River Watershed Stations. Source: http://or.water.usgs.gov/grapher/

Baseline Turbidity Levels in Oregon

In order to characterize natural turbidity patterns, DEQ examined data from several Oregon sites where the USGS has deployed continuous turbidity monitors. Data were divided into three "seasons" corresponding to fall/early winter (October-January), late winter and spring (February-May), and summer (June-September). Daily medians were tabulated by the USGS Data Grapher system (http://or.water.usgs.gov/cgi-bin/grapher/graph_setup.pl?basin_id=klamath#step1). DEQ calculated the median and 90th percentile for each season in which turbidity was available for at least 90% of days in the season. Results are presented in Figures 4-9. Table 1 provides a key showing the names of each station. Vertical lines represent the range of seasonal median/90th percentile turbidities for each station; the marker indicates the median season (for example, the median turbidity at the Rogue River station ranged from 0 to nearly 8 FNU among the 12 seasons of data available with a median of about 2 FNU). Numbers in parentheses after the name of each station represent the number of years in which sufficient turbidity data was available.

In the summer, median turbidity is extremely low (median 0-2 FNU with turbidity rarely getting above about 5 FNU), with the exception of the urban and agricultural watersheds of the Johnson Creek and Tualatin Rivers. In the wet season, median turbidity is typically very low (less than 2 NTU), even during rainy portions of the year. This seems to be particularly true in mountainous, forested watersheds, such as the North Umpqua, McKenzie, Clackamas, and North Santiam watersheds. At these stations, even 90th percentile turbidity in a season only occasionally reaches 15-20 FNU; for the most part, 90th percentile turbidity is 10 FNU or below. In these watersheds, the data indicate that, while turbidity levels rise with increasing flow during storm events, turbidity quickly subsides to baseline levels. This pattern also appears to be true at the Rogue River site (at Dodge Bridge), although this site is located in a valley and the surrounding land use is generally pasture/hay and cultivated crops (ODEQ 2008).[3] Lower turbidity in this region may be due to a general milder climate, more stable lithology, or other factors.

At the Williamson River site in the Upper Klamath Basin, seasonal median turbidity levels remained low during the dry summer months (0.7-1.7 FNU) and during the October-January (1.8-3.5 FNU) period. However, during the February-May period, median turbidity was generally much higher, ranging from 4.9-10.5 FNU, and 90th percentile turbidity generally ranged from 9-23 FNU. Spring coincides with the peak runoff period in this watershed. Spring flows in the Williamson River have increased significantly in the last century, possibly due to decreased evapo-transpiration due to riparian vegetation removal and wetland drainage in the basin, as well as increased snowmelt rates due to timber harvest in the upper Williamson basin (Risley and Laenen 1998).

The Johnson River site and the Tualatin River sites had higher seasonal median and 90th percentile turbidities than the other sites year round. All of these sites are located within the Willamette Valley ecoregion. With the possible exception of the Gales Creek site, these sites are located in areas that have intensive agricultural or urban land use. Baseline turbidity at these sites (during periods of low flow between storms) tends to be higher and turbidity at these sites tends to remain somewhat elevated even after discharge subsides after a rain event. It is unclear whether the generally higher baseline turbidity is due to a more erosive soil lithology, the more intensive land use patterns, or a combination of the two.

A six-year DEQ ambient monitoring study completed during the dry season in 2002 inventoried small wadeable stream sites in Oregon's eight ecoregions (Drake 2004). The study noted that overall median turbidity levels were approximately 1 NTU, regardless of lithology (resistant or erodible), or the degree of human disturbance. Reference site medians for all ecoregions were 1 NTU, except for the Willamette Valley ecoregion with a median of 2 NTUs. It should be noted that the Drake study did not examine wet season conditions, when higher levels of sediment-laden runoff from precipitation and snowmelt contribute to higher turbidity levels from natural and anthropogenic sources, nor did it examine background turbidities in higher order streams, which are of most interest for many point sources that discharge to these waters and could have permits with limits based on water quality standards.

Table 1. USGS stations used for seasonal turbidity analysis in Figures 3-8

Name	Station (Watershed)
Blowout C.	Blowout Creek (Santiam)
Clackamas Esta.	Clackamas River at Estacada (Clackamas)
Clackamas O.C.	Clackamas River at Oregon City (Clackamas)
Dairy Creek	Dairy Creek at Highway 8 (Tualatin)
Gales Creek	Gales Creek (Tualatin)
Johnson Creek	Johnson Creek at Regner Road (Johnson Creek)
L.N. Santiam	Little North Santiam River at Mehama (Santiam River)
McKenzie	McKenzie River above South Fork McKenzie River (McKenzie)
N. Santiam	North Santiam River at Mehama (Santiam)
N. Umpqua	North Umpqua River near Idleyld Park (Umpqua)
Rock Creek	Rock Creek (Tualatin)
Rogue	Rogue River at Dodge Bridge (Rogue)
SF McKenzie	South Fork McKenzie River near Rainbow (McKenzie)
Williamson	Williamson River at Chiloquin (Williamson)

[3] The Oregon water quality index has indicated that water quality in this area of the Rogue River is generally "excellent" (ODEQ 2008).

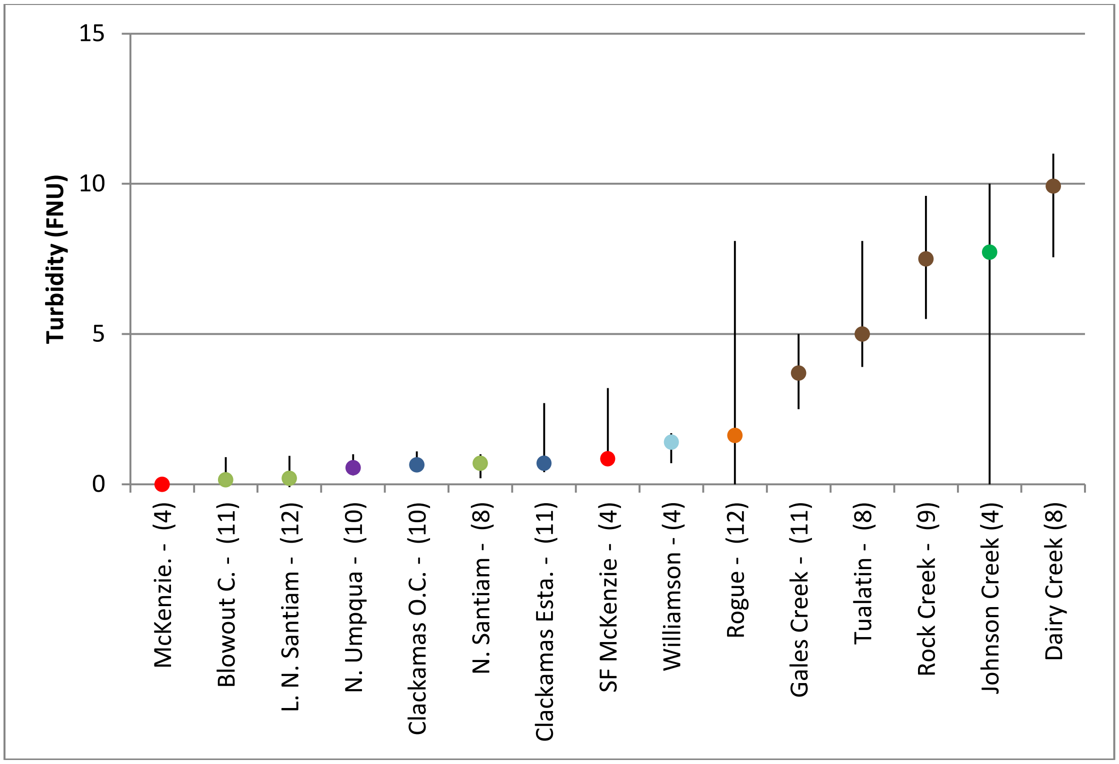

Figure 4. Summer (June-September) median turbidity (FNU) at USGS turbidity monitoring stations. (Data retrieved from http://or.water.usgs.gov/grapher/)

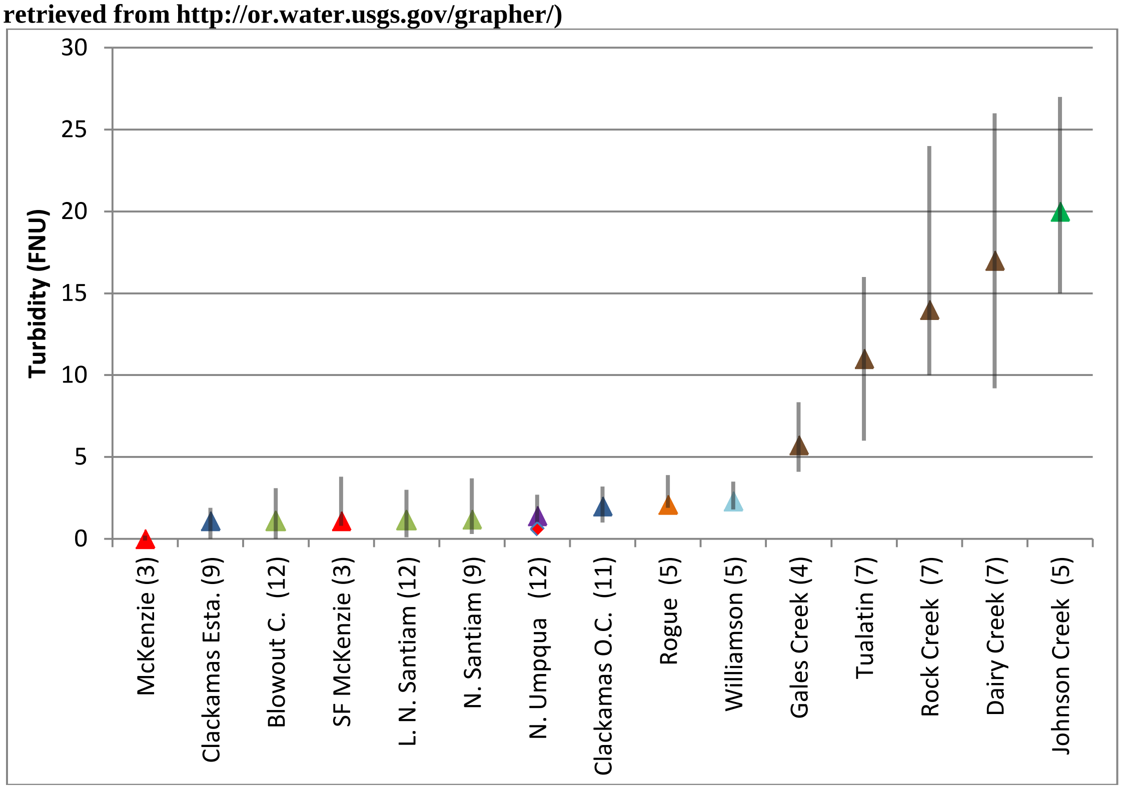

Figure 5. Fall/Winter (October-January) seasonal median turbidity (FNU) at USGS turbidity monitoring stations. (Data retrieved from http://or.water.usgs.gov/grapher/)

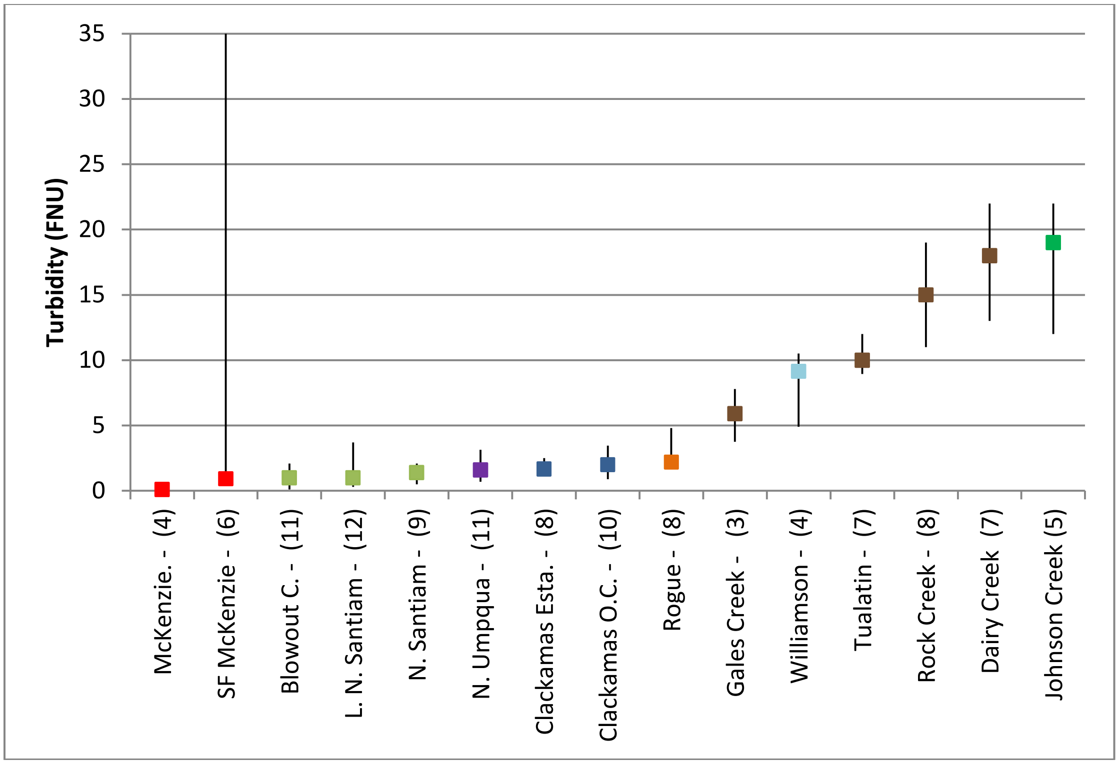

Figure 6. Winter/Spring (February-May) seasonal median turbidity (FNU) at USGS turbidity monitoring stations. (Data retrieved from http://or.water.usgs.gov/grapher/)

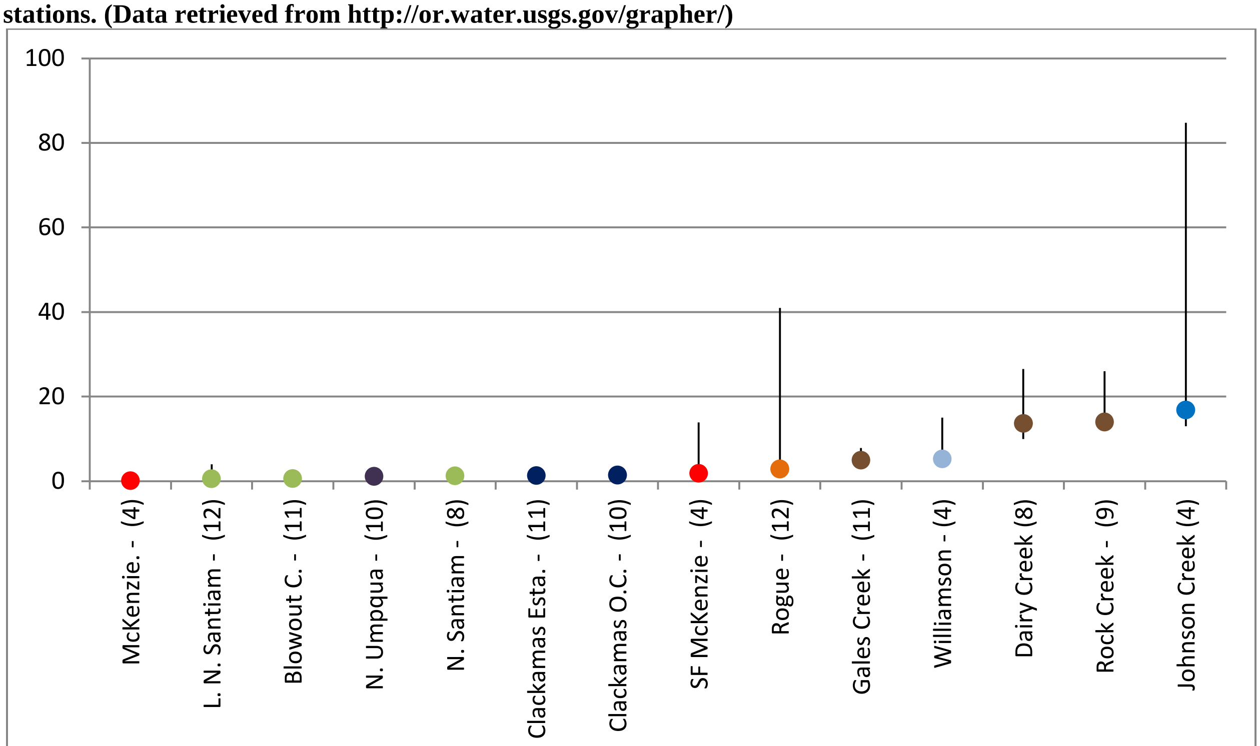

Figure 7. Summer (June-September) 90th percentile turbidity (FNU) at USGS turbidity monitoring stations. (Data retrieved from http://or.water.usgs.gov/grapher/)

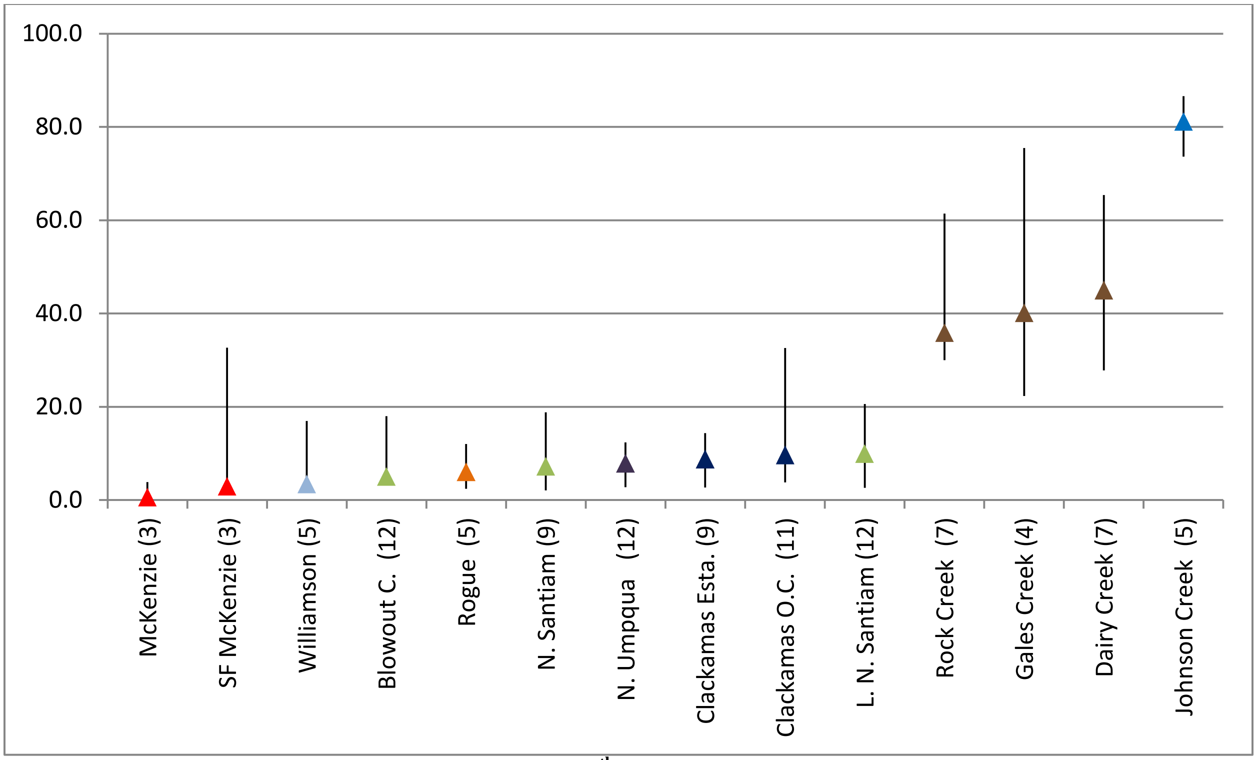

Figure 8. Fall/Winter (October-January) seasonal 90th percentile turbidity (FNU) at USGS turbidity monitoring stations.

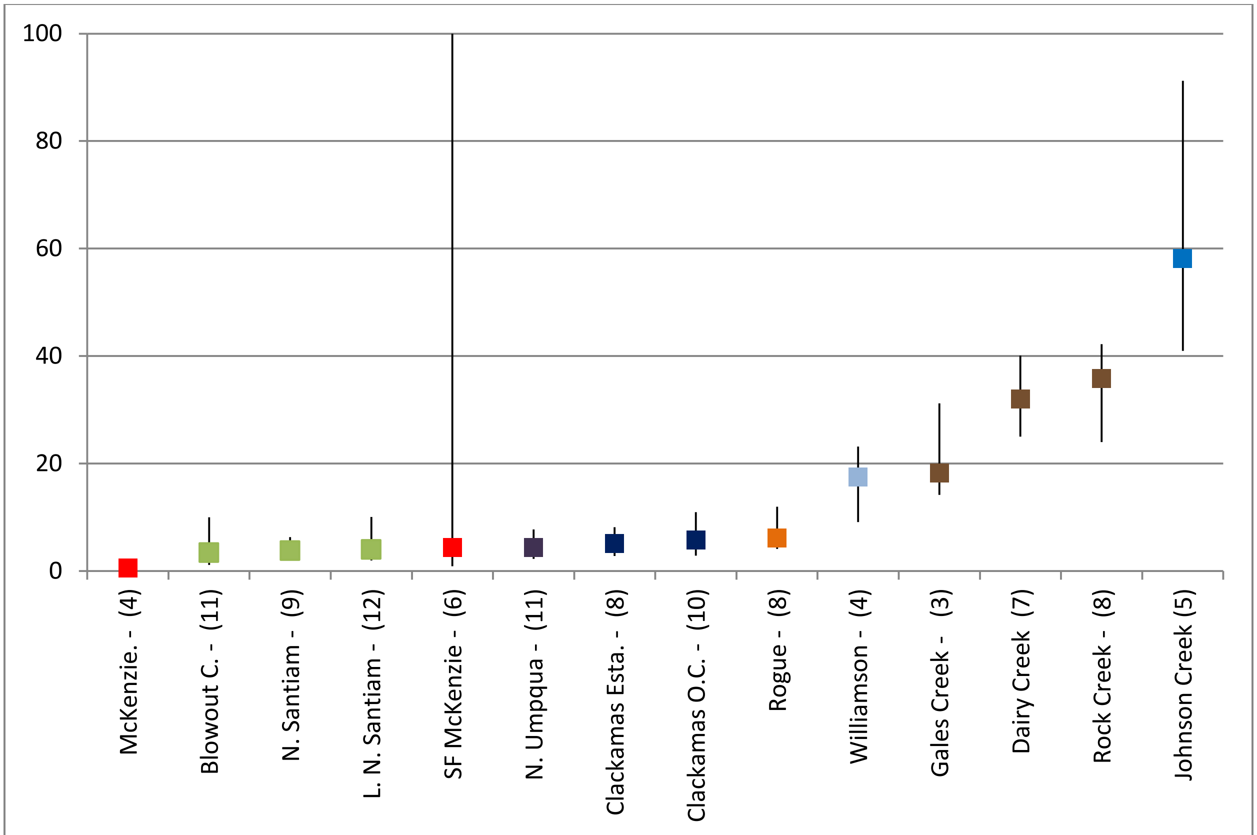

Figure 9. Winter/Spring (February-May) 90th percentile turbidity (FNU) at USGS turbidity monitoring stations. (Data retrieved from http://or.water.usgs.gov/grapher/)

Turbidity in Hood River and Other Glacial-dominated Streams

In stream systems dominated by glacial meltwater, such as the Hood River, the turbidity regime may differ significantly from those dominated by rainwater. In the Hood River, turbidity peaks are highest in the summer months due to the predominance of glacial till (Bonnie Lamb, ODEQ, pers. comm.) Turbidity often shows a diurnal cycle during the summer with peak turbidity occurring with daily peak flows, which occur in the afternoon due to increased ice and snow melt, as indicated by data provided by the Middle Fork Irrigation District (Figure 10). Further downstream, the Hood River is a mix of glacial- and storm-dominated systems. In this portion of the river, summer turbidity levels are elevated compared to wet season turbidity (Table 2). DEQ does not have sufficient data to determine whether this pattern is found in other glacial-dominated systems in Oregon.

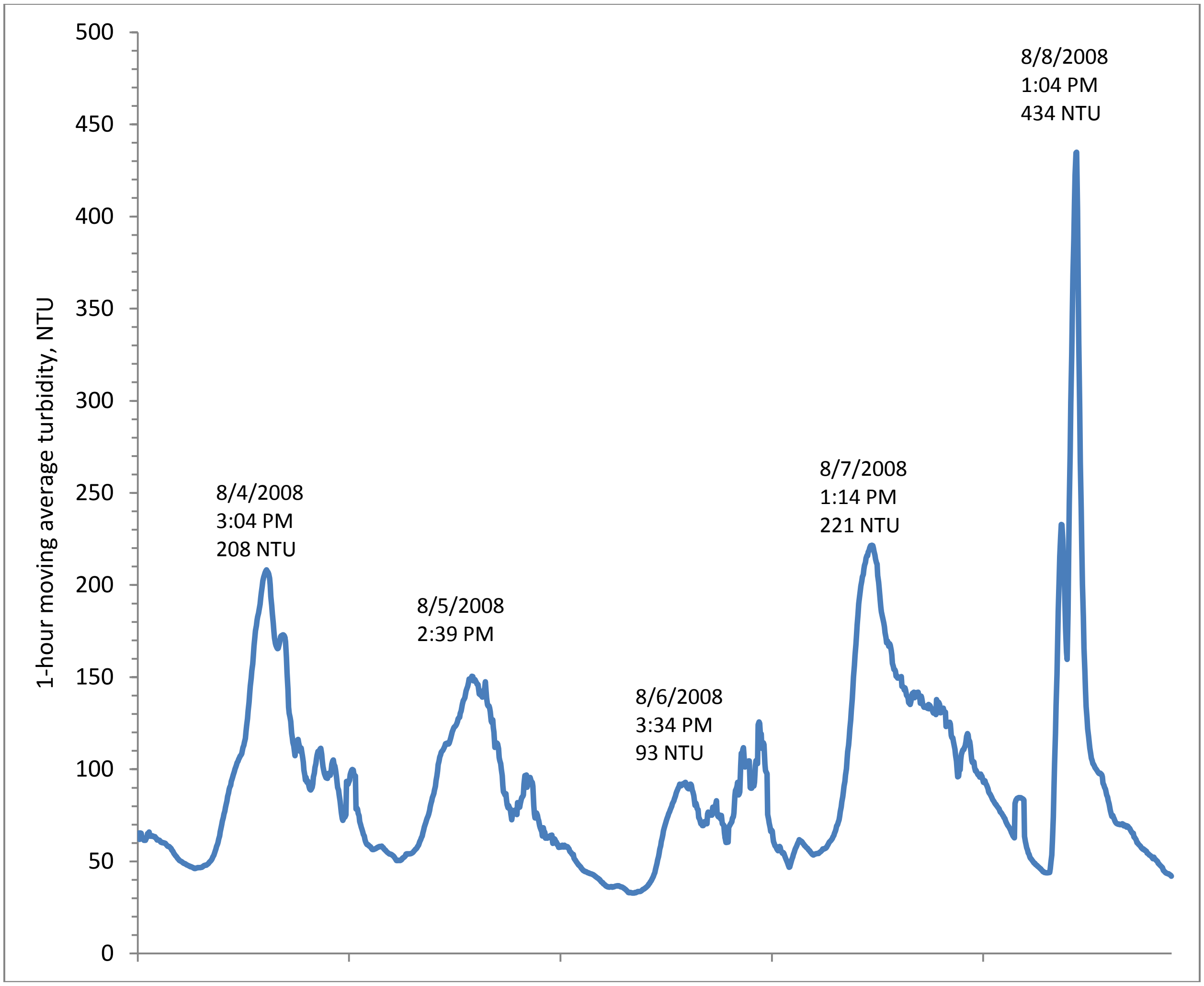

Figure 10. Turbidity in Middle Fork Hood River (near Parkdale, OR) from 8/4/2008 to 8/9/2008. Data provided by Middle Fork Irrigation District.

Table 2. Median Turbidity (NTU) at DEQ Ambient Sites, 1979-2002.

Site	**October-May**	**June-September**
Hood River at I84 (in Hood River)	3.5	5
Hood River at Highway 35 (in Hood River)	2	6
Hood River at Tucker Bridge (north of Odell)	2	7

Concentration-Duration-Frequency Analysis

Turbidity effects on aquatic life and drinking water treatability often are a function of duration, frequency, and turbidity level. A“concentration-duration-frequency” (CDF) analysis is a useful way to examine turbidity patterns (Schwartz, et al. 2008). A CDF analysis analyzes the frequency at which a particular “concentration” (e.g., turbidity level) is exceeded for a particular duration. CDF analyses are useful in examining data to determine if turbidity is exceeding “concentration/duration” thresholds that would be expected to result in an adverse effect on beneficial uses. A disadvantage, however, is that CDF anlyses effectively “decouple” the sequencing of turbidity events in time and thus may decrease the capability of understanding temporal variations in turbidity (e.g., seasonal patterns) or the potential role of antropogenic factors (e.g., land use practices).

DEQ conducted CDF analyses using turbidity data from continuous monitoring stations operated and maintained by the U.S. Geological Survey (USGS) and available at the USGS Oregon Water Science Center (http://or.water.usgs.gov/grapher/). For the stations examined, turbidity readings are reported in Formazin Nephelometric Units (FNUs). For the South Fork McKenzie River and Rogue River stations, readings were taken every 30 minutes. For the Beaverton Creek station, readings were taken every hour. For the analysis, DEQ utilized three stations highlighting three different turbidity regimes. The first station is on the South Fork McKenzie River, just above Cougar Lake. This station is in a forested area devoid of anthropogenic influences (C. Anderson, *pers. comm*.) The second station is on Beaverton Creek at 170th Avenue in Beaverton, Oregon. This station shows a turbidity regime in a small stream with urban land use. The third station is on Rogue River at Dodge Bridge, near Eagle Point, Oregon. This station represents a turbidity regime in a large-order stream with agricultural and rural residential land use.

DEQ examined three years of turbidity data from each site to conduct the analysis. Ideally, data from the same three years could be utilized; unfortunately, due to a large amount of missing data during various periods, DEQ had to utilize different periods for each station:

- South Fork McKenzie River Station (10/1/2003-9/30/2006)
- Beaverton Creek Station (10/1/2006-9/30/2009)
- Rogue River (3/9/1998-3/8/2001)

DEQ compared turbidity readings at the stations to thresholds of 5, 20, 55, and 150 FNU[4]. The 5 NTU threshold was chosen due to reported effects on drinking water treatability at this level, as discussed in Chapter 4. The remaining levels were chosen based on aquatic life thresholds described in Newcombe (2003). In the analysis an "event" is described as any reading or continuous series of readings of the applicable turbidity threshold, with the following decision criteria:

- Events exceeding 5 FNU for only one reading were not counted, as these could be due to debris passing by the optical sensor or another inaccuracy (Schwartz, et al. 2008).
- "Events" separated by only one reading were combined and counted as one event.

CDF curves for the three sites are presented in Figure 11 and Table 3. The graphs show the number of "events" (continuous exceedances) lasting a given duration or longer. The table provides a summary of events at specific turbidity/duration combinations for comparison sake. For example, at the South Fork McKenzie River, there were nine events exceeding 20 FNU that lasted 1.5 hours or longer and two events exceeding 150 FNU that lasted 5 hours or longer. The longest 5 FNU "event" at the South Fork McKenzie was nearly five days (119.5 hours), the longest 20 FNU event was 53.5 hours, the longest 55 FNU event was 36.5 hours, and the longest 150 FNU event was 10.5 hours.

[4] DEQ did not count 5 NTU events at the Beaverton Creek site, as nearly 80% of readings at that site exceeded 5 NTU.

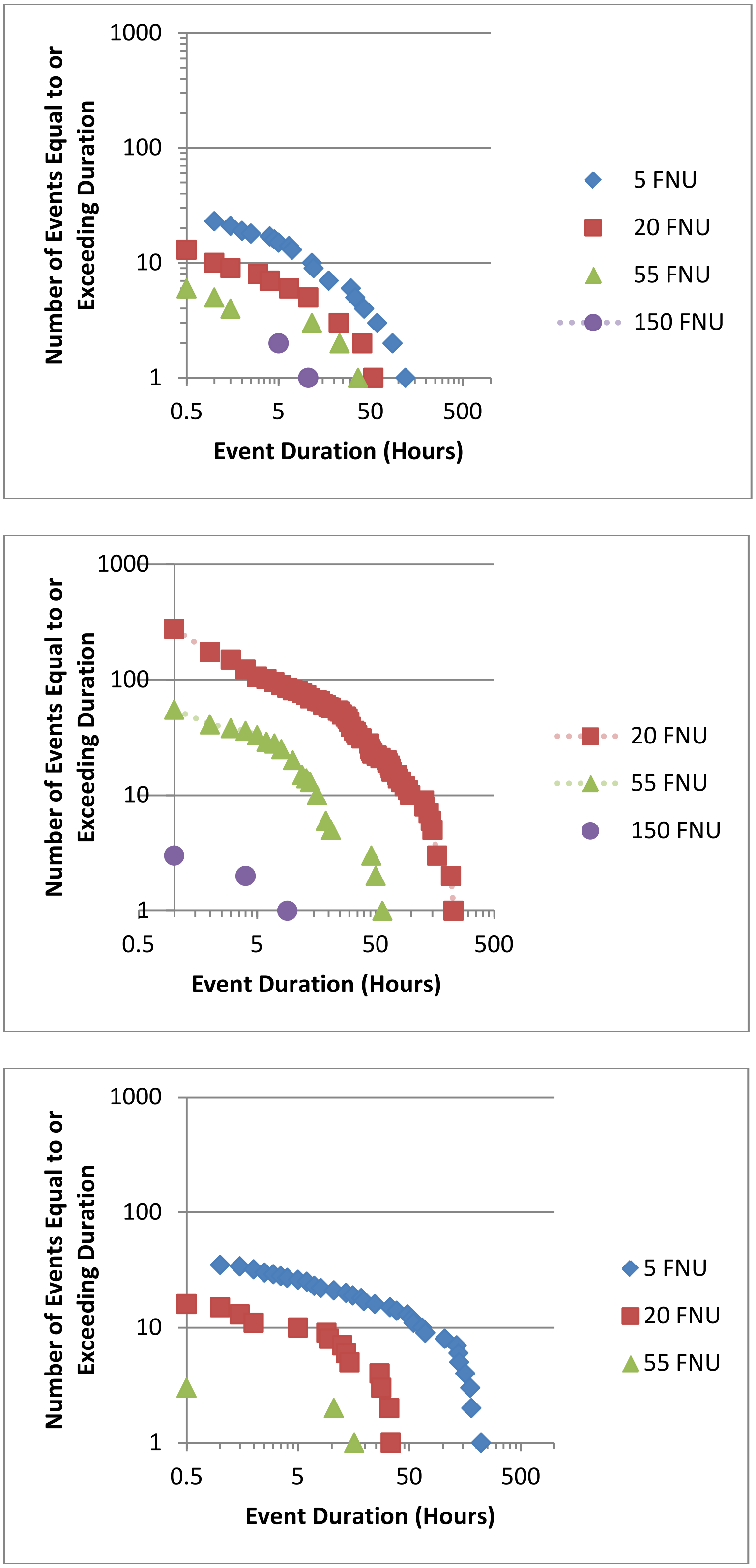

Figure 11. CDF Curves for USGS Turbidity data from South Fork McKenzie River above Cougar Lake (10/1/2003-9/30/2006), Beaverton Creek at 170th Ave., Beaverton, Oregon (10/1/2006-9/30/2009), and Dodge Bridge near Eagle Point, Oregon (3/9/1998-3/8/2001). Data from USGS Oregon Water Science Center, http://or.water.usgs.gov/grapher/.

Table 3. Turbidity "concentration/duration" exceedance events at three USGS stations.

Concentration/ Duration Combination	South Fork McKenzie River	Beaverton Creek	Rogue River
5 FNU/96 hours	1	Not analyzed	8
20 FNU/72 hours	0	15	0
55 FNU/12 hours	2	15	1
150 FNU/6 hours	1	1	0

The CDF analysis underscores some key differences in the different watershed types. The South Fork McKenzie analysis highlighted how, in many well-protected forested streams, turbidity readings tend to return to baseline relatively quickly during events (i.e., high turbidity (55 NTU, 150 NTU) events tend to be infrequent and short-lived). In contrast, Beaverton Creek has a "flashy" turbidity pattern in which small increases in flow, most likely from storm runoff, results in increased turbidity, which often persists for days or weeks. For example, at the South Fork McKenzie station, in three years, there were only 10 events exceeding 20 FNU lasting an hour or more with the longest lasting just over two days (53.5 hours). By contrast, at the Beaverton station, there were 275 events exceeding 20 FNU and lasting an hour or more; the longest lasted nearly 5 days (227 hours).

Turbidity at the Rogue River station exhibited an entirely different pattern with about the same number of "events" at each threshold as the South Fork McKenzie station. However, the events, particularly at the 5 and 20 FNU thresholds, tended to last longer. For example, there were seven 5 FNU "events" at the Rogue River station lasting longer than 133 hours; the longest 5 FNU event at the South Fork McKenzie station was 119.5 hours. This pattern would be expected in a higher order valley stream, due to slower settling time of finer sediments and the fact that a larger stream tends to average out effects of its tributaries and larger particles are depositived when river gradient decreases, resulting in fewer events of very high turbidity.

Another potentially useful way to compare the stations is to determine how often continuous turbidity readings exceed a given "concentration/duration" threshold that would expect to have an effect on beneficial uses. For example, Newcombe (2003) modeled that exposure to approximately 20 NTU for 150 hours would result in "moderately serious" effects to fish (likely, reduced feeding efficiency). This concentration/duration threshold was exceeded five times over three years at the Beaverton station and not at all at the other two stations. Exposure to 55 NTU for 24 hours would result in the same effect level. This C/D level was exceeded once at the South Fork McKenzie station (although another event lasted 23 hours), five times at the Beaverton Creek station, and not at all at the Rogue River station.

Longitudinal Patterns in Turbidity

Along the course of a waterbody, turbidity may increase or decrease due to a number of factors. Dams, inputs of clear water from tributaries, and settling of solids may decrease turbidity in a stream. Resuspension, inputs of turbid water, erosion and anthropogenic inputs may increase turbidity.

Once in the system, turbidity-causing materials may be conserved in the water column, deposited in the channel, washed out into the flood plain, or transported downstream. Subsequent high flow events can re-suspend turbidity-causing sediments into the water column. Larger, heavier particles tend to settle first, while smaller clay particles remain suspended for a longer period of time, contributing to downstream turbidity levels.

Only a handful of studies have examined longitudinal changes in turbidity along the course of water bodies in Oregon. Hughes and Gammon (1987) measured turbidity and other parameters two times at each of 26 sites along the mainstem Willamette River in August 1983 to examine the interaction of fish assemblage data and water quality. The data show turbidity peaks associated with a wastewater treatment plant (at river kilometer (RK) 283), a pulp-and-paper mill (RK 232), a landfill (RK 137), and a natural slough (RK 93) (Figure 12). Disregarding these data, turbidity appears to decrease from RK 283 to about RK 150, then gradually increase from RK 150 to the mouth. This is only a snapshot of the Willamette in one month and is limited as to its applicability to other locations and time periods.

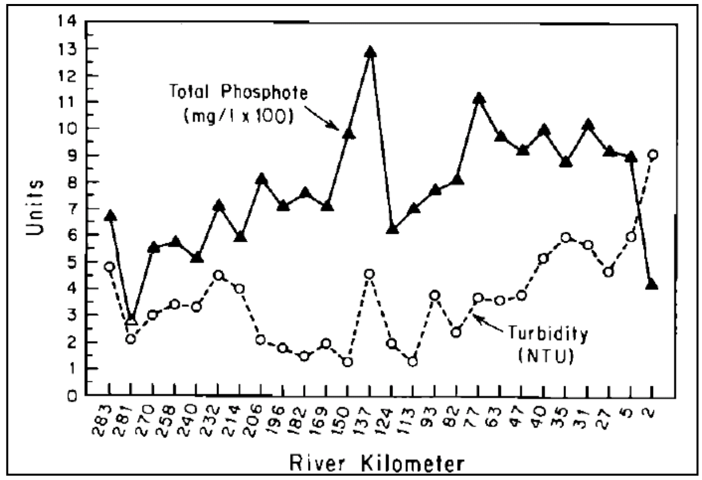

Figure 12. Median turbidity at 26 Willamette River sites in August 1983. Source: Hughes and Gammon (1987).

The National Council for Air and Stream Improvement (NCASI), as part of a Long-Term Receiving Water Study (LTRWS) has collected turbidity data periodically since 1997 at four sites on each of the McKenzie and Willamette Rivers (NCASI 2002). DEQ examined 1997-2009 NCASI data at these sites. Data were categorized as “dry season” (June-September), “early wet season” (October-January) and “late wet season” (February-May). The McKenzie River data (Figure 13) indicates that the turbidity trends slightly higher from upstream to downstream in the wet season and generally level during the summer. The Willamette River data (Figure 14) indicate upward trends in turbidity from upstream to downstream during the early and late wet seasons, but no consistent trend during the dry season. Turbidity readings in both rivers, and, in particular, the Willamette, could be influenced by a number of factors, such as storm water runoff, natural settling and resuspension, dams, effluent discharges, and inputs from tributaries.

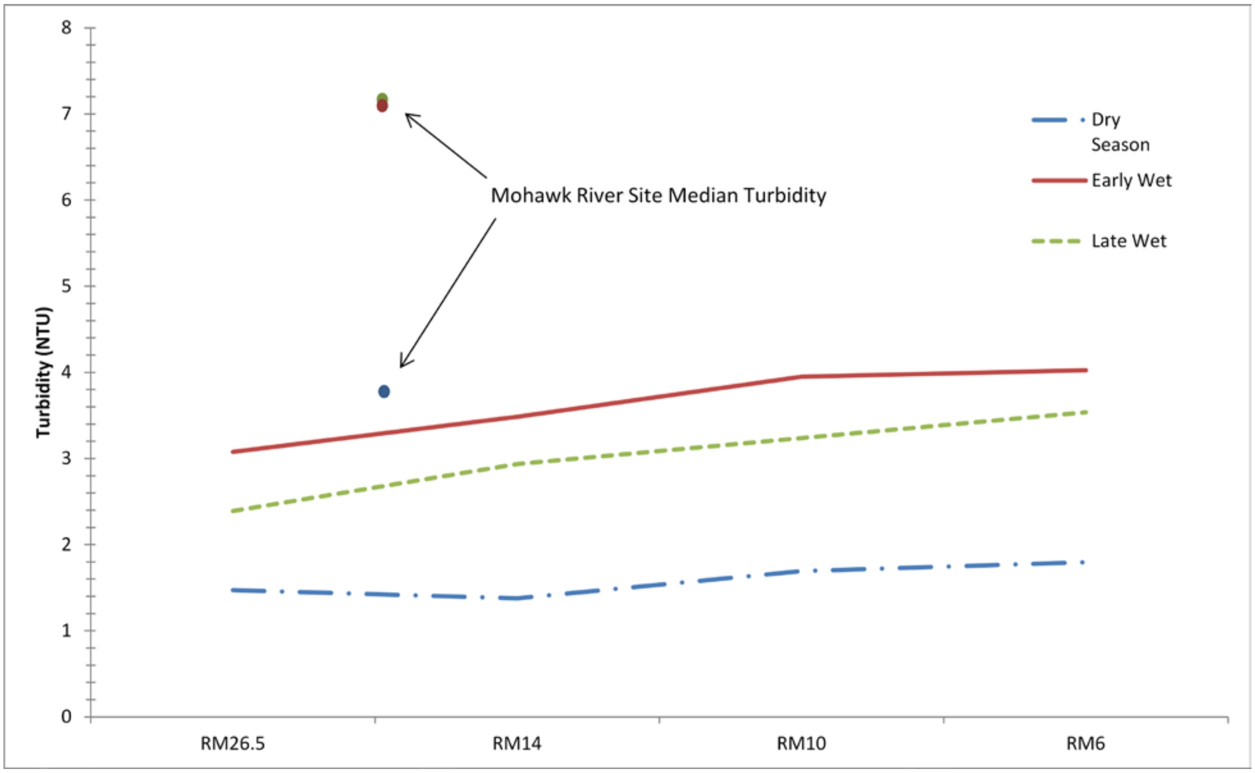

Figure 13. Median turbidity (NTU) at four sites on the McKenzie River, OR, 1997-2009. Data provided by NCASI.

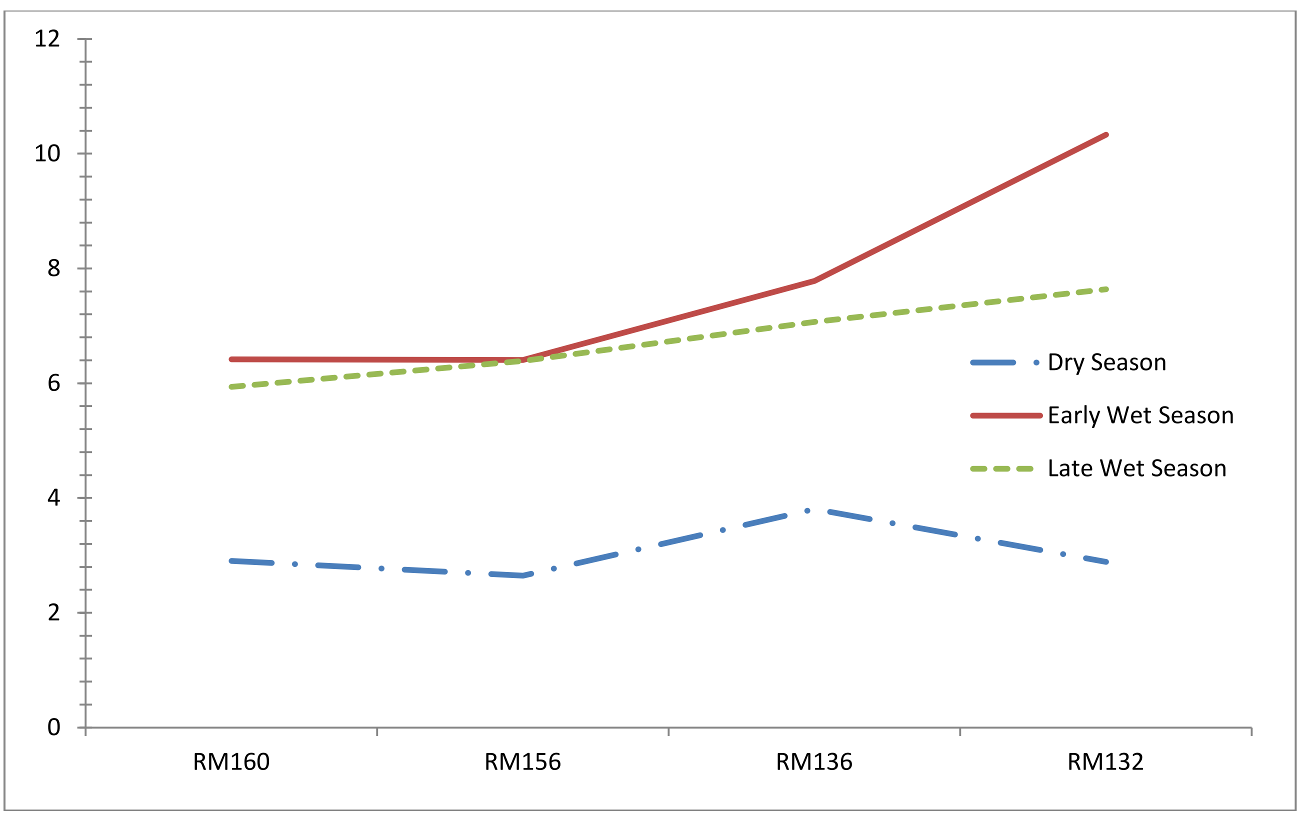

Figure 14. Median turbidity (NTU) at four sites on the Willamette River, OR, 1997-2009. Data provided by NCASI.

Sources of Increased Turbidity

Natural and anthropogenic inputs of sediments, particulate organic matter, and dissolved organic matter into the water column can result in increased turbidity levels. Algae, whether natural or induced by anthropogenic nutrient inputs, also can increase turbidity levels, but to a lesser extent than suspended sediments. Major factors controlling turbidity magnitude, duration, frequency and composition include precipitation, stream gradient, geology, natural disturbance and land use, all of which can be highly variable. Land use practices and wildfires, particularly preceding large storms, can result in massive inputs of turbidity-causing sediment to stream channels (May and Lee 2004).

Various types of land uses, both contemporary and historical, can result in increased turbidity, especially if best management practices (BMPs) have been poorly implemented. In agricultural and grazing areas, removal of vegetation and compacting of soil can cause increased runoff that carries eroded topsoil into rivers. Improper application of fertilizer may increase loads of nutrients that result in turbid algal blooms. Moreover, erosion of stream beds and banks that are destabilized through removal of vegetation or altered hydrology can contribute to turbidity. In areas with forestry operations, timber-harvesting practices, road construction, slash disposal, and site preparation can increase inputs of turbidity-causing sediment to streams. Urbanization prevents rain from penetrating into the soil, which increases surface runoff and results in transport of soil into streams directly or in stormwater outfalls. Erosion of soils at construction sites without proper controls can result in turbidity-causing soil loadings. Placer mining operations expose soils and can result in chronic turbidity issues. Industrial effluents and stormwater directly input turbidity-laden water into streams. Once sediment settles out of water, activities such as dredging without proper controls can re-suspend fine sediments, which may persist in the water column in some conditions.

Appendix A provides a summary of literature regarding sources of increased turbidity.

Conclusions

The information examined here highlights the difficulty in characterizing natural turbidity regimes in Oregon. However, some general conclusions can be made. During the dry summer season (June-September), Oregon streams tend to have low median turbidity levels. Headwater and wadable streams are typically less than 3 NTU and even larger order valley bottom streams tend to be less than 5 NTU during the summer months.

During the winter season, in mountainous forested streams, turbidity tends to be low except during storm events, when turbidity may increase sharply and peak before or at the same time as peak. Turbidity generally returns to baseline within 72 hours of peak flow. However, in areas that are prone to higher erosion, turbidity can persist for longer. At the same time, even in watersheds/regions with high clay content, the transport of sediment will likely be a hydrologically-driven phenomenon (Beschta, 1987). In larger order streams, such as the Willamette and Rogue Rivers, turbidity is generally less responsive to short storm events, tending to rise and ebb more slowly due to basinwide patterns of precipitation.

In streams where flow is dominated by glacial or snow melt, data indicate turbidity patterns are significantly different than in rain-dominated forested areas. In the upper reaches of such streams, turbidity may peak during the summer months, when glacial melt carries large amounts of sediment, such as in Hood River, or may peak in the spring, as is the case in the Williamson River (although the latter case also may be affected by anthropogenic influences). In addition, turbidity may exhibit a diurnal pattern during times of rapid snowmelt, peaking in the late afternoon/early evening in response to warmer temperatures, and decreasing as temperatures decrease.

The data also indicate that high intensity land use results in flashier and higher turbidity. For example, data from Johnson Creek and the Tualatin River watersheds had turbidity that was higher than mountainous regions; the higher turbdities tend to remain high even after flow subsides. More analysis would have to be done to tease out additional anthropogenic and natural factors contributing to turbidity in these watersheds.

Chapter 3. Effects of Increased Turbidity on Aquatic Life

Effects of increased turbidity levels on aquatic life vary with the magnitude of turbidity, duration and frequency of exposure, the physical characteristics of the material and other factors. These factors can result in decreased clarity and affect the sensitivity of the organisms present in a body of water (for example, some fish have been shown to be more sensitive to turbid conditions than others).

There are hundreds of studies describing the effects of turbidity and reduced light penetration and visual clarity on aquatic life. Thus, it was necessary to come up with a way to determine those studies most relevant to setting a water quality standard for turbidity. A greater weight of evidence is placed on studies having the following attributes:

- The research reported turbidity levels measured, the number of samples taken, and the duration over which the population of interest was exposed to turbidity.
- The research included control populations.
- The research included appropriate statistical analyses.
- Laboratory experiments must sufficiently mirror real world settings in order to make any extrapolations realistic. In particular, this qualifier is important for fish feeding studies, in which prey availability is an important variable.

In addition, a greater weight of evidence has been placed on studies that have been conducted in the Pacific Northwest including Washington, Oregon, Idaho, British Columbia and northern California. However, in some cases, studies from elsewhere have been included.

- In considering studies on the effects of turbidity on primary productivity and invertebrates, only a few qualifying studies were found in the Pacific Northwest. DEQ found other studies examining such effects in Alaska and New Zealand. While our review found that there are site-specific differences in the relationship between turbidity and primary productivity, Lloyd, et al. (1987) found that effects of turbidity on primary productivity in Alaska was likely applicable to forested streams of the Pacific Northwest. In addition, climatic conditions in New Zealand may be similar to Oregon and the country has a comparable array of streams and lakes, making extrapolation possible (R. Peterson, *pers. comm..)*; as such, such studies were considered.

- Literature regarding effects of turbidity on submerged aquatic vegetation in estuaries relies primarily on investigations conducted on the West Coast. While this topic has been studied in depth on the east coast of the United States, especially in the Chesapeake Bay, and in northern Europe, Thom, et al. (2008) and others note that systems in the Pacific Northwest differ substantially from those areas, due to differences in tidal ranges and regimes. As a result, DEQ focused its review to literature on West Coast estuarine effects.

- Fish-effects literature examines effects of turbidity on both native Oregon fish species and recreationally important non-native fish species that are present in Oregon. Water quality standards are designed to protect a broad range of aquatic organisms, and DEQ accordingly included a broad array of fish-effects literature. In its review, DEQ highlighted the potential effects of turbidity on native species, in addition to its review of potential effects to other species of fish.

Other literature is also cited, primarily to describe generally the mechanism by which turbidity affects the various endpoints.

DEQ presented effects separately for streams, lakes, and estuaries to determine if separate water quality criteria are necessary to protect aquatic life for each type of ecosystem. A summary table of reported literature is provided at the end of the streams and lakes/reservoirs sections (pages 33 and 41, respectively). The tables present effects in order of increasing turbidity measurements. In addition, DEQ presented information regarding duration of exposure for each study (e.g., chronic, 5 days, 1 hour, etc.), as effects of short, but sharp increases in turbidity levels are expected to be different from those of chronic, low level turbidity increases. Such information will assist DEQ in developing water quality criteria that include magnitude, duration, and frequency considerations. The summary table also notes whether each study was conducted in the laboratory or the field, and what instrument was used to measure turbidity, if reported at all.

DEQ did not provide a summary table for the estuary section as the different metrics used to describe turbidity effects in estuaries (e.g., irradiance, attenuation coefficient, NTU, suspended sediment concentration) makes it impractical to compare studies to one another.

Effects of turbidity on primary productivity

Summary

Primary productivity (the growth of periphyton, phytoplankton and macrophytes) provides the base of the food chain in aquatic systems, influencing food available for aquatic invertebrates and fish. Primary productivity depends on the availability of light in the water body to fuel photosynthesis. Increased turbidity levels can decrease available light in the water column, potentially decreasing productivity, which, in turn may reduce food availability for higher trophic levels (Sorensen 1977). The USEPA based its recommended 1976 turbidity criteria, which is purportedly the basis for Oregon's current standard, on how turbidity effects primary productivity. Specifically, EPA's criterion focuses on the effects of turbidity on the "compensation point," or the depth in the water column at which the rate of photosynthesis is equivalent to the rate of respiration. EPA recommended the following criterion:

> "The combined effect of color and turbidity should not change the compensation point more than 10 percent from its seasonally established norm, nor should such a change place more than 10% of the biomass of photosynthetic organisms below the compensation point." (EPA 1976)

Studies show that increased turbidity in streams, lakes and estuaries reduces various measures of primary productivity, such as benthic algal production and the presence and growth of various macrophytes. The level of turbidity which results in reduced productivity seems to vary considerably, however; some studies indicate that small increases in productivity (6 NTU) reduce productivity; other studies found that even very large levels of turbidity did not reduce productivity substantially. Aquatic plants partially compensate for reduced productivity by increasing photosynthetic efficiency, at least for some period; however, this may result in a cost to overall growth. The extent to which productivity is affected can vary depending on the type of sediment, depth of stream, water color, nutrient levels and endpoint examined in the study. For example, clay particles tend to attenuate light more efficiently than larger particles.

Modeled relationships between turbidity, light extinction, and productivity indicate that small increases in turbidity of less than 5 NTU would result in minor (3-13%) reduction in benthic algal productivity in stream 0.5 meters deep. The lowest empirically measured effect level was approximately a 6 NTU increase in turbidity in streams (Davies-Colley, et al. (1992) and ponds (Reed, et al. 1983). However, studies of macrophytes in streams did not detect significant differences in productivity except at much higher levels 100 NTU and higher (Parkhill and Gulliver 2002). In general, while available studies do show a pattern between increased turbidity and algal-based chlorophyll, there are certain concerns with each study that would make it difficult to determine turbidity levels and durations that would equate to negative effects. Moreover, the lack of multi-year studies on effect of increased turbidity on primary productivity (and corresponding secondary productivity effects) is a major data gap, particularly with respect to determining whether turbidity is resulting in impaired aquatic life uses. Finally, irrespective of concerns related to individual studies, there was no consistent pattern between specific turbidity levels and reductions in primary productivity that are conducive to developing a water quality standard.

In estuaries, considerable research has been conducted on the effect of reduced light penetration on presence of eelgrass. In the Yaquina Bay estuary, Brown, et al. (2007) have recommended water quality criteria expressed as a light extinction coefficient. However, there is as yet insufficient data to determine similar criteria for other estuaries in Oregon.

Literature regarding effects of turbidity on primary productivity - streams

The ability of light to penetrate through water depends upon the irradiance or reflectivity of the water surface, the absorption of light by color and the reflection and absorption of light by particles and other matter in the water column. Light penetration through water is represented by the Beer-Lambert law: $I_z=I_0e^{-Kd*z}$; where I_o is the irradiance at the water surface, I_z is the irradiance with a penetration of light at depth = z, and where the light attenuation with depth is related to the light attenuation rate, K_d (m^{-1}). With respect to productivity, K_d is termed the photosynthetically active radiation (PAR) attenuation coefficient, which is the vertical attenuation rate for the photosynthetic waveband (400 nm-700 nm) (Kirk 1985). The proportion of light making it through water is dependent on K_d, which is itself affected by turbidity.

The relationship between increased turbidity and reduced light penetration is curvilinear, but varies by stream or even by storm event. Van Nieuwenhuyse (1983) related light penetration to turbidity in placer-mined Alaska streams using the equation $I_z=10^{(2.00-NtZ)}$, where N_t was the total extinction coefficient and was related to turbidity (NTU) according to the equation $N_t= 1.00 + 0.024*T$. Parkhill and Gulliver (2002) developed a similar equation for experimental streams in Minnesota: $N_t = 2.619 + 0.129*T$ (Figure 15). The relationship developed in the Alaska study was for a compilation of several streams; individually, each stream may have had fairly different relationships between compensation point and turbidity.

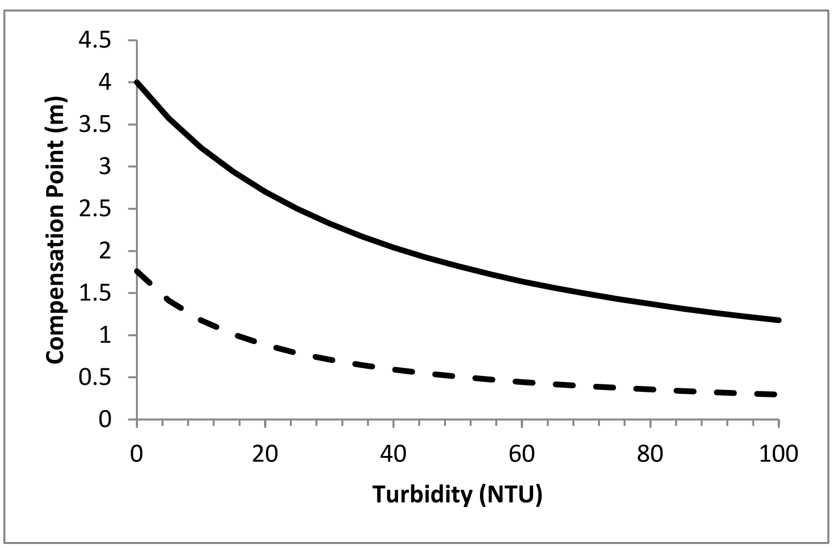

Figure 15. Modeled relationship of turbidity to compensation point in Alaskan glacial streams and experimental streams in Minnesota.

A few studies have compared turbidity and primary productivity rates in streams. Van Nieuwenhuyse and LaPerriere (1986), in studying moderately- and heavily-mined streams in Alaska over two open water seasons (June-October), found a significant linear relationship between productivity and incident PAR, itself dependent on turbidity; however, turbidity levels in mined streams were significantly higher than generally measured in Oregon and the differences in productivity between unmined streams were greater than between mined sites within one stream. In two similar streams, productivity measured over ten days in the moderately mined turbid stream (average turbidity 170 NTU ± 60 NTU; mean depth 0.22 m) was about half with respect to the unmined stream (average turbidity 0.73 ± 0.26 NTU; mean depth 0.31 m), although photosynthetic efficiency was about double in the turbid stream. In the same stream, mean chlorophyll *a* density on an artificial substrate was about 60% less in the mined stream as compared to the unmined stream after 16 weeks of exposure.

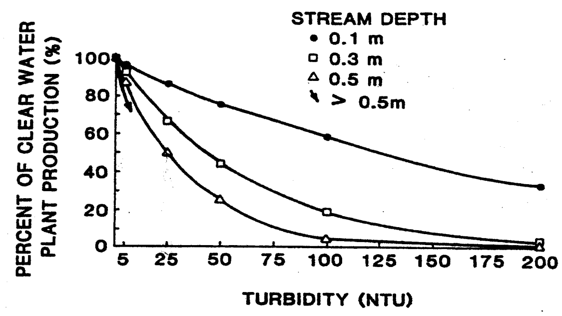

FIGURE 5.—Reduction in plant production with increased turbidity in artificially turbid streams of various depths in interior Alaska (derived from equation 7); NTU is nephelometric turbidity unit.

Figure 16. Modeled relationship of turbidity and primary production in Alaskan glacial streams. Figure 5 in Lloyd, et al. (1987)

Using the relationship between turbidity and compensation point derived in Van Nieuwenhuyse and LaPerriere (1986), Lloyd, et al. (1987) modeled how turbidity would affect gross primary production in streams of various depths. This relationship indicated that a 5 NTU increase in turbidity in shallow, clear-water streams could potentially decrease primary productivity in clear streams by 3-13%, and a 25 NTU increase could decrease primary productivity in clear streams by 13-50% (Figure 16). Negative effects on primary production in streams were predicted to be larger at depths of greater than 0.5 meters. The authors cautioned that the model was conservative (i.e., understated effects) because the light extinction coefficient in clear water was greater than had been measured in clear water elsewhere.

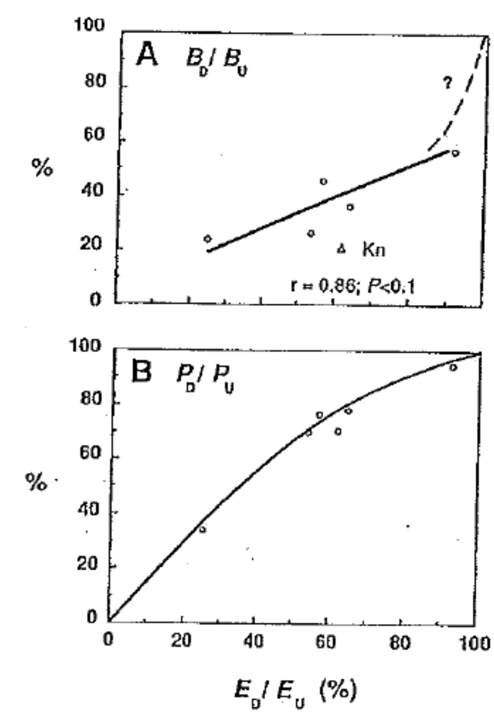

Figure 17. Relationship of light reduction and benthic algal production (top) and productivity (bottom) upstream and downstream of mining activities in New Zealand streams. (Figure 8, Davies-Colley, et al. 1992).

There are a number of issues that make it difficult to utilize the information from the Van Nieuwenhuyse and LaPerriere (1986) and Lloyd, et al. (1987) studies. For example, mined streams have high concentrations of iron, which may have affected the results. In addition, no error estimates are provides and it assumes ongoing turbidity conditions, which may not be a reasonable supposition in many cases (Flinders, *pers. comm.*, 12/17/13).

A two-month field study on New Zealand streams where placer mining was occurring showed that increased turbidity downstream of mining caused significant decreases in periphyton productivity and benthic algal biomass as compared to upstream of mining activities (Figure 17; Davies-Colley, et al. 1992). In the stream with the smallest upstream/downstream difference in median turbidity (median upstream turbidity 1.1 NTU; median downstream turbidity 7.3 NTU), benthic algal biomass decreased from 50.4 mg/m^2 to 12.8 mg/m^2. Overall, benthic algal biomass at downstream sites was 15-

57% of upstream biomass. In plotting light reduction against benthic algal biomass reduction and reduction in productivity, the authors found a significant relationship.

Peer reviewers to this document have raised concerns with the Davies-Colley study regarding its utility in developing a water quality standard for turbidity. This includes the high variability in both turbidity and periphyton production endpoints, the method and time frame by which productivity data were collected, the lack of control populations, the general variability of lotic systems making comparison of upstream and downstream sites difficult, and the nature of the streams studied (C. Flinders, *pers. comm.,* 12/17/13).

Parkhill and Gulliver (2002) found in a study of eight experimental streams in Minnesota that turbidity had little effect on daily photosynthetic production of a species of macrophyte (measured as *chlorophyll a*), but did affect whole stream metabolism at 25-35 NTUs. The authors concluded that, while turbidity may affect autotrophic productivity less due to increased photosynthetic efficiency, even small loads of sediment in the system decreased overall biological activity in streams (Parkhill and Gulliver 2002). This is consistent with Odum (1985), who suggested that, while plants can adapt to higher turbidity levels, such an adaptation would divert energy from growth and production to maintenance.

Literature regarding effects of turbidity on primary productivity – lakes and reservoirs

Effects of turbidity on primary productivity in lakes, reservoirs, and ponds are similar to that in streams; however, in lentic waters, higher turbidity levels may persist for longer periods, exacerbating such effects and more clearly translating to reductions in secondary productivity. Unfortunately, while there is a relatively strong body of research documenting how increased turbidity and/or reduced light penetration impact phytoplankton and macrophytes, few papers connect specific nephelometric turbidity levels to impacts, making it difficult to extrapolate these studies to setting statewide turbidity criteria.

As in streams, turbidity in lakes reduce the volume of the water body in which photosynthesis can occur (Kirk 1985; Lloyd, et al. 1987). A seasonal 5 NTU increase in turbidity reduced the photosynthetically active volume of naturally clear lakes in Alaska by as much as 80% (Lloyd, et al. 1987). Koenings, et al. (1990) found significantly lower chlorophyll *a* levels in glacial lakes at 33 NTU than in clear or stained lakes, which the authors hypothesized was due to a combination of higher turbidity and lower temperature and food levels.

Shrader (2000) studied the interactions of turbidity, phosphorus, and productivity in Prineville Reservoir on the Crooked River and concluded that turbidity may significantly affect energy allocation and transfer between trophic levels (from phytoplankton to zooplankton to fish) in the Prineville Reservoir. The study also noted that phosphate adsorbing onto turbidity-causing clay particles might be partially responsible for decreased chlorophyll-*a* levels found in the reservoir, although this effect was minor with respect to the effects of decreased light in the water column.

Literature regarding effects of turbidity on primary productivity – estuaries

The sediment dynamics of estuaries, which affect turbidity and light levels, are extremely variable, particularly in the estuarine turbidity maximum, where the marine- and river-dominated portions of the estuaries combine. In these areas, tides force saline marine water beneath the fresh river water, resulting in high amounts of suspended sediment and a high degree of light attenuation depending on particle size (Cloern 1987; Campbell 1987). In the Columbia River Estuary, the position of the ETM and its concentration of suspended sediments can vary with tidal changes and volume of upstream discharge (Morgan 1992). In the Columbia, The turbidity maximum is generally most pronounced during summer low flow periods (Callaway, et al. 1988).

In many coastal plain estuaries, such as the Columbia River estuary, suspended sediment-caused turbidity limits phytoplankton production (Morgan 1992; Cloern 1987). Lara-Lara, et al. (1990) found a negative correlation between the daily phytoplankton production in the Columbia River estuary and the light extinction coefficient, although other factors (solar irradiance, temperature, chlorophyll *a* concentrations, and suspended sediment concentration) also affect phytoplankton production. However, populations of certain zooplankton are positively correlated with increased levels of suspended particulate matter in the Columbia River estuary (Morgan, et al. 1997) leading to overall higher secondary production.

Much of the literature examining water clarity in estuaries in Oregon and elsewhere focuses on the effect of light on algal growth and growth of submersed macrophytes (commonly referred in the literature as submerged aquatic vegetation, SAV). The literature places a particular focus on the effects of light attenuation on presence and growth of eelgrass, *Zostera marina.* Eelgrass serves as an important refuge for juvenile fish, protecting them from predation. Seagrass also moderates current velocity, increases water clarity by promoting sediment deposition, removes nutrients from the water column, and provides other environmental benefits (Brown, et al. 2007).

Reduced light penetration limits growth of SAV in estuaries and has exacerbated a decline in eelgrass around the world caused by anthropogenic nutrient inputs (Giesen, et al. 1990; Moore, et al. 1996). Goldsborough and Kemp (1988) found that a submerged macrophyte exposed to shaded conditions equaling 11% of ambient light for seventeen days experienced significant reductions in biomass and stem density; reproduction was eliminated entirely. Duarte (1991) suggested that coastal seagrasses require 11% of surface irradiance at the sea bottom in order to grow. U.S. EPA set water quality criteria for visual clarity in the Chesapeake Bay, which range from 0.2 to 1.9 meters Secchi Depth depending on the salinity regime and application depth (USEPA 2003). Batiuk, et al. (2000) recommended a water clarity criterion for SAV ranging from 15-22% of surface irradiance depending on salinity zones in the Chesapeake Bay.

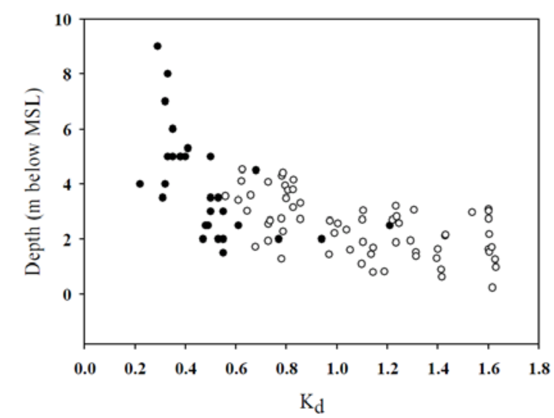

Figure 11.7 Relationship between *Z. marina* maximum depth limit (m below MSL) and k_d. Filled circles represent data from Duarte (1991) and hollow circles represent data from the Yaquina Estuary.

Figure 18. Relationship between eelgrass lower depth limit and light attenuation coefficient in the Yaquina estuary (Figure 11.7 in Brown, et al. 2007).

Key differences exist between estuarine systems in the Chesapeake Bay and those in Oregon that indicate that water clarity criteria designed to protect SAV in the former should not be extrapolated to the latter. These factors include differences in tidal ranges and regimes (Thom, et al. 2008), temperature (Boese, et al. 2009), and winter irradiance (Boese, et al. 2005). Brown, et al. (2007) recommended water clarity criteria (expressed as light attenuation coefficient) of 0.8 m^{-1} and 1.5 m^{-1} in the marine dominated and riverine-dominated portions of the Yaquina Bay Estuary, respectively. These limits are based on the relationship between light attenuation coefficient (K_d) and eelgrass lower depth limit, which is the lowest depth at which eelgrass will grow (Figure 18). Boese, et al. 2009 found a significant relationship between K_d and lower depth limit of eelgrass in the Yaquina Bay estuary, but not in six other Oregon estuaries, factors including current velocity, sediment characteristics, water temperature and salinity affected the eelgrass range. Additional information is needed on light gradients and SAV distributions in other estuaries, as well as seasonal patterns in light between estuaries and salinity (C. Brown, *personal communication*).

Sources of uncertainty regarding effects of turbidity on primary productivity

Sufficient data exist to indicate that reduced light penetration, whether due to inorganic or organic turbidity or shade, reduces the amount of light that can reach a given depth in streams, lakes, and estuaries, ultimately reducing productivity. Nevertheless, there are few studies that are useful in identifying specific turbidity levels at which effects are found and there is some discrepancy between the turbidity level that causes effects on benthic algae (e.g., Davies-Colley, et al. 1992) and levels that may affect macrophytes (Parkhill and Gulliver 2002); moreover, individual studies documented here each have limitations and, as a whole, there is no consistent relationship in considering the evidence as a whole that help to identify a specific turbidity level at which productivity decreases. Studies do show decreased productivity at approximately 6 NTU in some cases, but much smaller changes at turbidity levels of 170 NTU in alaska. Moreover, the literature tended to limit turbidity and productivity measurements to a few months. DEQ was unable to find literature documenting the impact of turbidity on primary productivity over several years of repeated exposure.

In lentic systems, no regional studies directly compared turbidity levels to measures of productivity, and that study was conducted in a shallow pond in North Carolina and measured only growth of a bottom macrophyte. As a result, there is considerable uncertainty with respect to specific turbidity levels that may correspond to reductions in turbidity in lakes.

In Oregon estuaries, the primary endpoint of concern with respect to turbidity is growth of eelgrass. However, only in Yaquina Bay has a relationship been well-established between light extinction and eelgrass presence. In other estuaries, research has not been able to show a clear relationship between turbidity and eelgrass presence due to the influence of other factors.

Effects of turbidity on macroinvertebrates and primary consumers

Summary

Increased turbidity is correlated with various metrics of decreased benthic macroinvertebrate abundance and diversity, as well as populations of other primary consumers. There are two ways in which turbidity may affect such populations: 1) turbidity may reduce food availability for primary consumers by limiting primary production and 2) increased turbidity and suspended sediment may increase drift of macroinvertebrates due to clogging of benthic habitat (Culp, et al. 1986).

There is very little literature examining the effect of turbidity on primary consumers communities in lakes and virtually none that would be relevant to lakes in Oregon. However, there are several studies examining the relationshiop of turbidity and suspended sediment with macroinvertebrate density and diversity in streams including DEQ’s own data, as well as other studies done using Oregon data. While there are concerns with designs of individual studies, the body of literature and data as a whole indicates that moderate levels of turbidity (4-8 NTU) in streams is correlated with a decrease in macroinvertebrate abundance and diversity indices.

Literature regarding effects of turbidity on macroinvertebrates - streams

Scherr, et al. (2011) noted that higher turbidity was significantly correlated with decreasing abundance of sediment-sensitive macroinvertebrate taxa in the Umatilla River, Oregon. Relatively low levels of turbidity, in the range of 4-8 NTU appeared to result in a notable decrease in abundance of the most sediment-sensitive taxon, the mayfly *Epeorus* and the water penny beetle, *Psephenus*, (Figure 19). Conductivity was found to be more important than turbidity in predicting invertebrate abundance, but turbidity also was significant, especially for *Epeorus* species. Peer reviewers have noted some concerns with this study regarding its utility for developing a water quality standard, including the lack of information regarding the turbidity levels measured, number of samples taken and duration of exposure, as well as the lack of control populations (Flinders, *pers. comm. 12/17/13)*

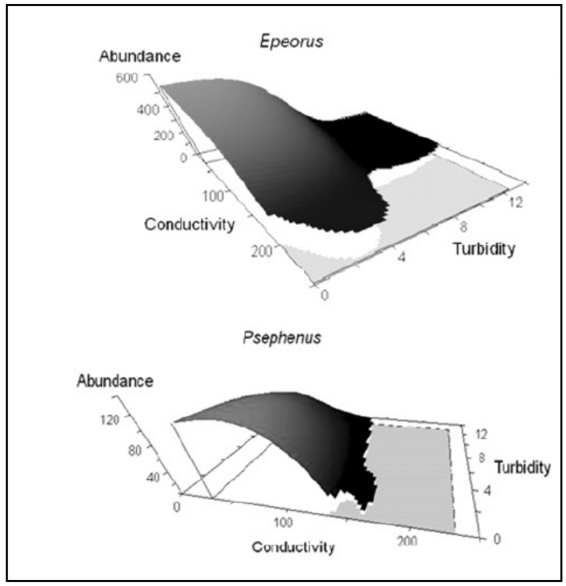

Figure 19. Relationship of turbidity and conductivity to abundance of two macro-invertebrate taxa in the Umatilla River, Oregon. (Figure 1 in Scherr, et al. (2011))

Quinn, et al. (1992) found that macroinvertebrate density decreased downstream of alluvial gold mining activities in six shallow (0.2-0.4 m) New Zealand streams. Macroinvertebrate density decreased by 50-80% when turbidity downstream of mining activities was 7-18 NTU higher than generally clear upstream conditions (Figure 20). Macroinvertebrate taxonomic richness decreased significantly between all but two of the upstream/downstream pairs. It's likely that turbidity effects would be different in deeper streams or systems with different plant and algal communities adapted to lower light conditions (IMST 2006). The Quinn, et al. (1992) study is a companion paper to the Davies-Colley, et al. (1992) study described in the primary productivity section. As a result, it is subject to similar concerns described above, in that results may be influenced by upstream-downstream differences in stream conditions other than turbidity (C. Flinders, *pers. comm*, 12/17/13).

Shaw and Richardson (2001) found in an experimental stream in British Columbia that sediment pulses of 23 NTU for nine days and 19 days decreased abundance and species richness of benthic macroinvertebrates, and that such decreases were more prominent as the length of the pulse increased.

DEQ and Unpublished Data

Data collected and analyzed by DEQ indicate a negative correlation between turbidity and macroinvertebrate density and diversity. In a 2007 report, DEQ analyzed chemical and biological data from 118 perennial and wadeable streams in Oregon. (Hubler 2007b). As part of the report, DEQ conducted a relative risk analysis to examine the likelihood that "poor" turbidity values[5] resulted in poor

[5] DEQ classified turbidity values as "good," "fair," and "poor" using level 3 ecoregion-specific turbidity reference conditions (Hubler 2007b). "Good" scores are the 75th percentile or less of reference sites in the ecoregion; "poor" scores are the 95th percentile or greater. Good and poor turbidity values are as follows:

Ecoregion	**Coast Range**		**Willamette Valley + Puget Lowlands**		**Willamette Valley**		**Cascades, East Cascades, and Blue Mountains**		**Klamath Mountains**		**Columbia Plateau, Northern Basin and Range and Snake River Plains**	
	Good	Poor	Good	Poor	Good	Poor	Good	Poor	Good	Poor	Good	Poor
Turbidity	<1	>6	<5	>30	<6	>22	<1	>2	<1	>3	<4	>13

macroinvertebrate conditions as compared to “good” turbidity values. The analysis showed that sites with high turbidity scores were 4 times as likely to have poor macroinvertebrate conditions as compared to those with good turbidity values with a lower 95% confidence level of 2.5. While the paired data approach may not indicate overall environmental conditions over weeks and months prior to sampling that could impact biological conditions (C. Flinders, *pers. comm.*, 12/17/13), the results suggest that biological conditions that are more frequently impaired when a stream is transporting sediments/solids during periods when they should be stable and not transporting as much sediment.

In an unpublished review of 1994-1995 EMAP data from 1st-3rd order streams in western Oregon, U.S. EPA researcher noted that there was an 85% chance of a stream being impacted (defined as an EPT index <18) at 4.4 NTU or higher. (John Paul, *unpub. data*).

DEQ compared observed biotic integrity measurement with observed winter turbidity measurement in a study of 27 first to third order coast ecoregion streams (Mulvey and Hamel 1998). The study used continuous turbidity measurements at four north coast streams during two storm events and at three mid coast streams during one storm event. Grab samples were taken 1-3 times at 20 additional locations. Macroinvertebrate data was taken the following summer. Results from continuous data indicate a significant relationship between the maximum turbidity during storm events and decreased macroinvertebrate and vertebrate indices (riffle score, pool score, and vertebrate index of biological intregrity) for two of the three storms. Grab samples indicate a significant relationship between maximum turbidity and decreases in all three indices in Mid Coast streams. In North Coast streams, there was a strong relationship between maximum turbidity and decreased riffle macroinvertebrate scores, but not pool macroinvertebrate scores or vertebrate IBI scores. While it is difficult to utilize these data for the purpose of developing a turbidity standard, it does indicate that streams with higher winter turbidities tended to show lower biological conditions, even when measured the following summer.

Relationship between EPT health and fish health

Taken together, the information presented in this section indicates that measures of macroinvertebrate health appear to be affected when turbidity is 4-10 NTU. However, the studies measuring effects around 4 NTU (Scherr, et al. 2011 and John Paul, *unpub. data*) focus on measurements of the health of ephemeroptera, plecopteria, and trichoptera, which are generally sensitive to suspended and bedded sediment and other pollutants. Studies that have found effects at 7 or 8 NTU (DEQ data and Quinn, et al. 1992) examined overall macroinvertebrate density. A question that remains is the extent to which effects on EPT species may result in effects on higher trophic levels, particularly fish. DEQ examined its own database for sites where there is paired data on metrics of EPT abundance and diversity, as well as fish biointegrity index (IBI). Fish IBI is measured using the methods in Hughes, et al. (2004). There is a significant but weak correlation between EPT abundance and fish IBI (Figure 21; r^2=0. 13) and an equally weak (r^2=0. 13) correlation between EPT diversity and fish IBI. There also is a slightly smaller correlation (r^2=0. 10) between % EPT and fish IBI. As a result, it is hard to say for certain whether turbidity, at levels that would affect sensitive macroinvertebrates, also would affect fish populations.

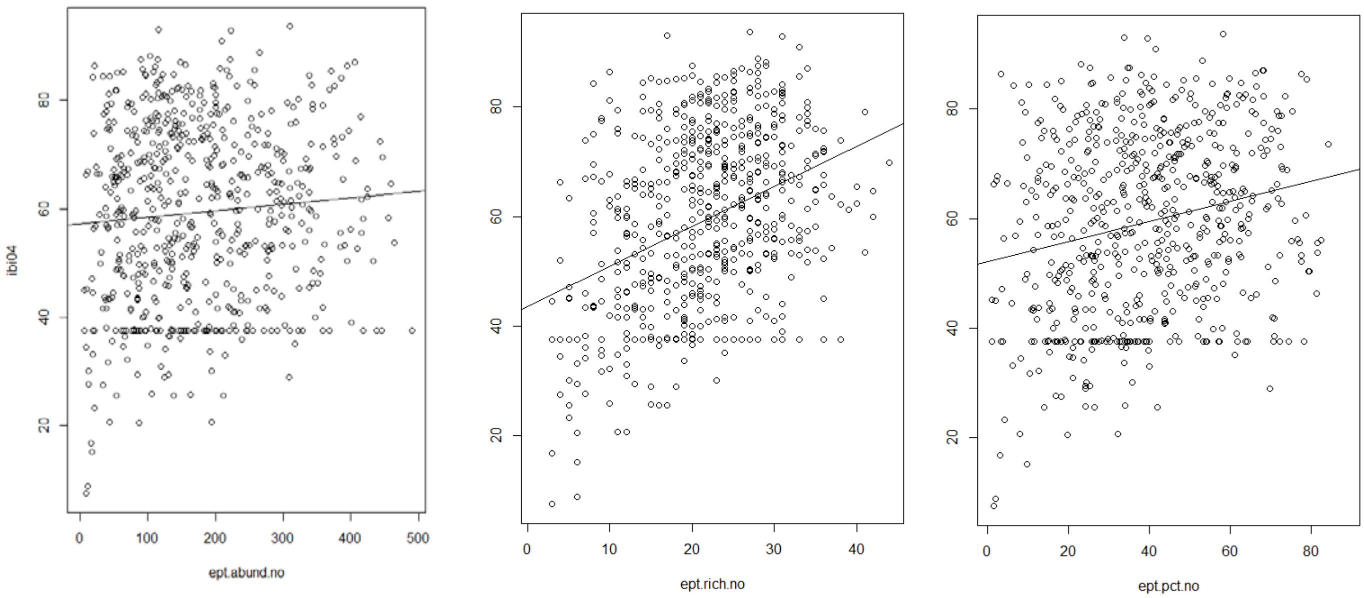

Figure 20. Fish IBI vs. EPT abundance (left), richness (center), and % EPT (right) in wadeable Oregon streams. (Hubler, *pers. comm.* September 10, 2013)

Literature regarding effects of turbidity on zooplankton - lakes

The body of literature useful to connecting specific turbidity levels to invertebrate health in lakes is sparse. Lloyd et al. (1987) noted that turbid glacial lakes in Alaska exhibited less than 5% of the zooplankton densities often associated with clear lakes; however, the study does not report specific turbidity levels that correspond to such effects; moreover, conditions of lakes in Alaska may be sufficiently different from those in Oregon to make any extrapolation difficult. Some literature suggests that increased turbidity is beneficial to large zooplankton due to decreases in susceptibility to visually searching predators (e.g., Fiksen and Giske 1995).

Sources of uncertainty

Specific concerns with the methodology of studies cited in this section are noted within the discussion of each study. Despite these concerns, the studies are relatively consistent in their findings that turbidity in streams from 4-10 NTU are associated with decreased macroinvertebrate abundance and diversity as compared to clear streams. One general source of uncertainty comes from the lack of longer terms studies examining impacts of chronic exposure to turbidity. The Prussian, et al. (1999) study starts to address that question in examining impacts of suction dredge mining a year after operation, but that study does not include multiple measurements of turbidity to impacts. The Quinn, et al. (1992) study measured impacts of increased turbidity on macroinvertebrates over two months and at a minimum turbidity increase of 7 NTU. Studies over a longer period and smaller turbidity increases could help to determine the lowest effects threshold. Another source of uncertainty is the extent to which impacts on invertebrates, in general, and EPT invertebrates, in particular, translate to overall ecosystem health. As noted, studies showing impacts to invertebrates around 4 NTU focused on EPT species, which those focusing on general macroinvertebrate indicators detected negative effects starting around 7 or 8 NTU.

As noted above, studies of turbidity in lakes and their impacts on zooplankton and other primary consumers is lacking, especially those that could be relevant to lakes in Oregon.

Effects of turbidity on fish

Summary

Turbidity may result in or be correlated to a number of different effects on fish. Such effects include:

- Behavioral and physiological effects
- Reduction of food abundance and availability
- Effects on prey detection, feeding success, and growth
- Increased cover for prey species

Studies examining behavioral and physiological effects of turbidity on salmon have examined avoidance, migration within the water column, territoriality, and blood chemistry. Avoidance behavior has been noted at 70 NTU in coho salmon. However, such behavior was noted in 30 minute trials. As a result, it is unclear if such fish would exhibit avoidance behavior over a longer time period at lesser turbidities, or if acclimation would occur.

There are numerous studies examining the effects of turbidity on prey detection (generally, in terms of "reactive distance," the distance at which visually-oriented fish physically aligns itself toward its prey), feeding success, and growth. In general, changes in reactive distance are noticeable at low levels of turbidity. Evidence of effects on feeding success are found at approximately 20 NTU in many salmonids, although fish that are accustomed to higher background turbidity are better adapted to higher turbidity levels. Studies examining these effects, while providing a basis for examining the effects of turbidity on fish growth, are not relevant to setting a water quality standard for turbidity, as they occur at turbidity levels and durations (minutes to hours) that frequently occur in undisturbed streams in Oregon and which do not represent a long-term threat to aquatic life in Oregon.

For this report, DEQ focused on those studies examining the effect of turbidity on growth of fish present in Oregon, as DEQ considers such studies as being relevant to setting a water quality standard to protect aquatic life.

Literature regarding effects of turbidity on fish in streams

Behavioral and physiological effects

A number of studies have examined the effects of high turbidity on fish behavior and physiology, focusing on salmonids. While the effects are indicative of how salmonids behave when exposed to higher turbidity, the studies are commonly limited to shorter trials and to turbidities and durations found in undisturbed watersheds. Moreover, the effects are not necessarily relevant to longer term impacts that could affect fish growth or populations.

A few studies have examined whether turbid water elicits an "avoidance" response in salmonids. In laboratory trials, juvenile coho salmon acclimated to clear water exhibited a significant avoidance response to suspended sediment at ~ 70 NTUs in 30 minute trials. Similar test fish that were acclimated to more turbid water conditions (2 - 15 NTUs) exhibited significant avoidance response at ~ 100 NTUs (Bisson and Bilby 1982). Brook trout from the Nemadji River, Wisconsin showed no preference for moderately (5.8 NTU) or highly turbid (56 NTU) water in two day trials (Gradall and Swenson 1982). Servizi and Martens (1992) found that coho salmon exposed to a gradient of suspended sediment preferred the surface, where suspended sediment concentrations were lower.

Berg (1982) conducted several experiments to examine the effects of turbidity pulses on different aspects of juvenile coho salmon behavior. The study found that juvenile coho exposed to pulses of 60 NTU spent significantly more time on the bottom substrate than in the water column (3-day exposure, although movement to the substrate was reported about 4 hours after exposure). Fish returned to the water column as turbidity levels were lowered to 10-20 NTU. Territorial behavior decreased and activity levels increased in 60 NTU turbidity pulses (3 days) then began to return to normal levels at 10 NTU. Turbid (20 and 60 NTU) treatments also increased the frequency of collisions between fish. Berg and Northcote (1985) performed similar studies on juvenile coho and found that territoriality and aggression decreased when exposed to 30 NTU for an hour as compared to those in clear water.

Servizi and Martens (1992) conducted several studies on sublethal responses of coho salmon to suspended sediments from the Fraser River. Mean blood sugar levels, a secondary stress indicator, increased positively with turbidity, with 25% and 50% increases corresponding to turbidities of 42 and 80 NTU, respectively.

Effects of turbidity on prey detection, feeding, and growth

Increased turbidity levels reduce the range at which visually-oriented fish can detect a contrast between a prey item and its surroundings. This effect reduces the distance at which these fish can detect their prey, called the reactive distance, which can reduce foraging success, growth rate, and long-term survival assuming constant food concentrations (Utne-Palm 2002). In addition, as visibility decreases due to increasing turbidity, piscivorous fish change from passive to active feeding strategies, potentially resulting in decreased growth rate due to extra energy expenditure (Sweka and Hartman 2001b).

In the context of setting water quality criteria to protect aquatic life, the pertinent questions in examining effects of increased turbidity on fish are, “What constitutes an effect?” and “What level and duration of turbidity are sufficient to cause such an effect?” A good way to illustrate such effects is using an inverted pyramid (Figure 23). At the base of the pyramid are studies examining reductions in reactive distance. Such effects do not depend on duration. The next level corresponds to turbidity levels/exposure durations that result in decreased foraging rates. These studies generally (but not exclusively) look at moderate turbidity levels (30 NTU and higher) with short durations (a few minutes to a few hours). At the next level are studies examining how turbidity may affect growth rates in fish. Such studies are generally conducted at low-to-moderate turbidity levels (10-20 NTU) over a period of several days to a few weeks. At the top level are studies that model the effect of increased turbidity over a season or longer to estimate effects on fish populations. However, empirical evidence of how turbidity impacts fish populations are virtually absent. At best, the Lloyd, et al. (1987) study from Alaska reports how turbidity affects coho salmon populations in glacial lakes in Alaska; however, even this study does not have sufficient information to connect specific turbidity levels to effects. At best, modeling techniques (Harvey and Railsback 2004; 2009) indicate how turbidity could impact salmonid

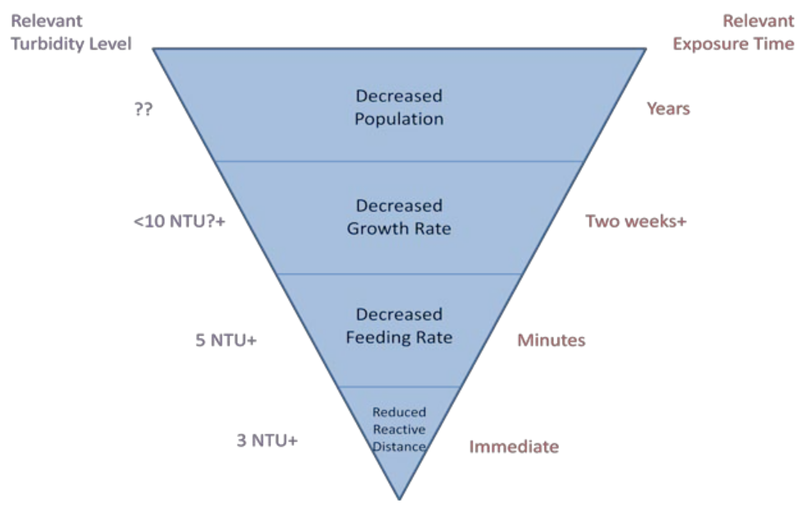

Figure 21. Schematic of effects of turbidity on fish feeding.

populations; however, to date such studies have mainly revealed key uncertainties about the critical connection between turbidity and food acquisition.

Reactive Distance Studies

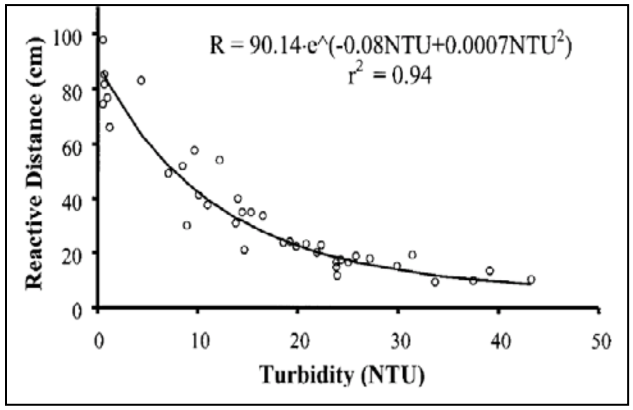

Figure 22. Relationship of reactive distance and turbidity for brook trout. Figure 1 in Sweka and Hartman (2001a).

Studies indicate that reactive distances of adult and juvenile fish decrease exponentially with increasing turbidity levels. Sweka and Hartman (2001a) found that, compared to clear water, reactive distance of brook trout decreased by 50% at 10 NTU (Figure 24), but noted that there was considerable variability in effects of turbidity on reactive distance below about 5 NTU. At the same time, once a fish had reacted to its prey, the probability of capture was not affected by turbidity. Barrett, et al. (1992) found that reactive distance of rainbow trout exposed to 15- and 30-NTU was 80% and 45%, respectively, of those exposed to ambient turbidities of 4-6 NTU. Berg and Northcote (1985) found that reactive distance in juvenile coho salmon decreased from 30 cm to 10 cm in a turbidity pulse of 60 NTU and did not recover to normal levels when turbidity decreased to 0 NTUs post-treatment.

Feeding Studies

A number of studies have shown that increased turbidity levels result in decreased feeding rates in. Berg (1982) found that juvenile coho salmon consumed significantly less prey when exposed to a turbidity pulse that was initially at 60 NTU for 3 days and gradually decreased over an additional 3 days. Prey consumption was depressed even as turbidity decreased from 60 NTU to 10 NTU. Juvenile coho also had more mis-strikes at 10 NTU in Berg's experiment and their response time was significantly higher at 10, 20, and 60 NTU. Berg and Northcote (1985) found that prey capture success was significantly reduced at 20, 30, and 60 NTU; most prey at these turbidity levels were captured downstream of the captor, whereas they were almost exclusively captured upstream in clear water conditions.

Other studies have shown that fish will feed in moderate turbidity, although feeding strategies and location may change. Feeding success of rainbow trout and coastal cutthroat trout collected from two northern California streams did not differ sharply for fish collected in high turbidity (66-317 NTU) and low turbidity (2-11 NTU) conditions (White and Harvey 2007). In a laboratory stream, Harvey and White (2008) tested the foraging success of juvenile cutthroat trout and coho salmon on drift and benthic prey at turbidities ranging from 0-400 NTU. Drift prey foraging success fell at 25 NTU and continued to decrease as turbidity levels increased. Benthic foraging success exceeded 50% up to 100 NTU and fell sharply at higher turbidities (Figure 25). The study also found that foraging success of cutthroat trout feeding on live, mobile oligochaetes was decreased at 50 NTU as compared to a clear water control. Gregory and Northcote (1993) and Gregory (1994) conducted experiments on feeding rates of small- and medium-sized juvenile Chinook salmon at different turbidities in an aquarium (i.e., standing water). In general, the salmonids had reduced foraging at 370 and 810 NTU. In clear water, foraging on benthic and surface prey was low, but was highest at 35-150 NTU. The salmon exhibited reduced feeding rates on surface *Drosophila* at all turbidity levels tested (18, 35, 70, 150, 370, and 810 NTU) with almost no feeding at 370 and 810 NTU. Large-sized juvenile Chinook maximized feeding on *Drosophila* at 150 and 370 NTU. Feeding on the plankton *Artemia* was generally not affected except at 370 and 810 NTU for all three size classes. Foraging on benthic *Tubifex* was highest for all size categories between 18 and 150 NTUs (Figure 26).

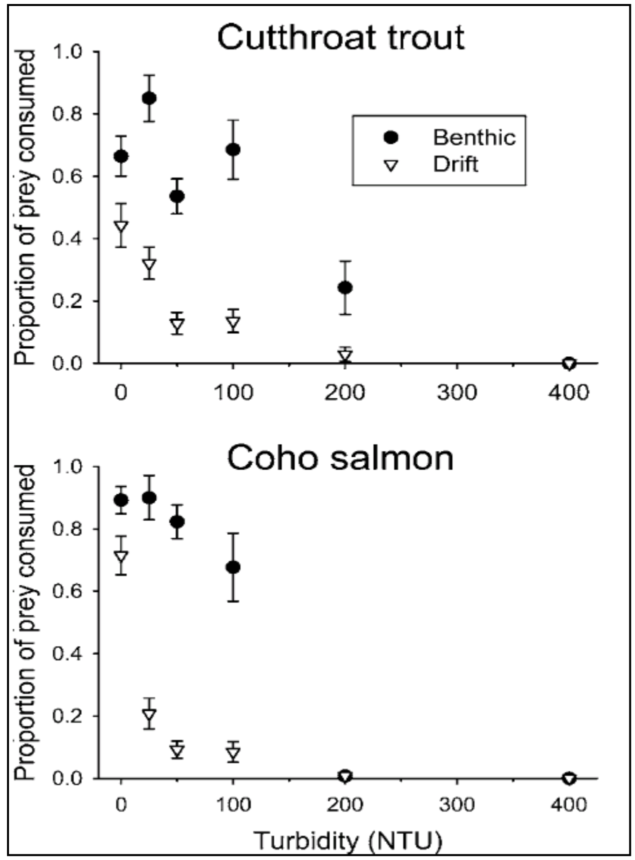

Figure 23. Turbidity's effects on foraging success in cutthroat trout and coho salmon. Figure 3 in Harvey and White (2008).

Growth Studies

A few studies have looked at how increased turbidity may affect growth rates in fish. Such effects may result from a combination of: 1) a reduced ability to detect prey; 2) a switch from a passive to an active foraging strategy, resulting in increased energy expenditure; and 3) reductions in food availability. Two studies focused on the first two of these effects without examining food availability reductions by controlling the rate of food introduced to the fish. In the first study, Sweka and Hartman (2001b) found that, even though mean daily consumption was unrelated to increased turbidity, specific growth rate was affected by increased turbidity, as fish used active foraging at higher turbidities, thus expending more energy. In that study, a linear relationship was developed between turbidity and growth rate in 5-day experiments on brook trout; the relationship corresponded to a 50% decrease in growth rate at 50 NTU. The study exposed fish to increasing levels of turbidity for five-day periods; as a result, there are potential issues with comparing experimental treatments to control groups. In addition, only a single prey type was used and only on the surface and supplied at a single rate. As a result, it is difficult to extrapolate the results of this study.

In the second study, Sigler, et al. (1984) examined how turbidity affected density and growth of newly emerged steelhead and coho salmon that were fed brine shrimp. Tests were done on fish exposed to various turbidity levels and durations in two oval channels and four raceway channels.[6] In a laboratory test in an oval channel, weight and length gains of newly emerged steelhead were significantly less in a 19 day exposure at 45 NTU compared to clear test waters. In a raceway channel, weight and length gains by steelhead were also reduced in a 19 day exposure at 38 NTU. Newly emerged coho salmon exposed to 22 NTU for 11 days in oval channels, and to 11-32 NTU for 14 days in raceway channels also experienced smaller length and weight increases during exposure. In raceway channels, biomass of juvenile Chinook was significantly reduced across the range of test levels (11-49 NTU) compared to the clear water control. The authors concluded that fish exposed to as little as 25 NTUs experienced decreasing growth. The data indicate that slightly smaller turbidity levels (22-23 NTU, the lowest levels measured) result in decreased growth in coho salmon and did not measure a constant turbidity level below that level. Thus, the data do not necessarily indicate a lowest effects level threshold; moreover, the experiment does not indicate if exposure to turbidity levels for longer than 11-21 days (one experiment was conducted for 31 days at 41 NTU) might result in decreased growth.

[6] In the oval channels, some experiments were conducted using water with two different turbidities (e.g., 143 NTU in one channel and 192 NTU in another); whereas others were conducted using clear water in one channel and turbid water in the other.

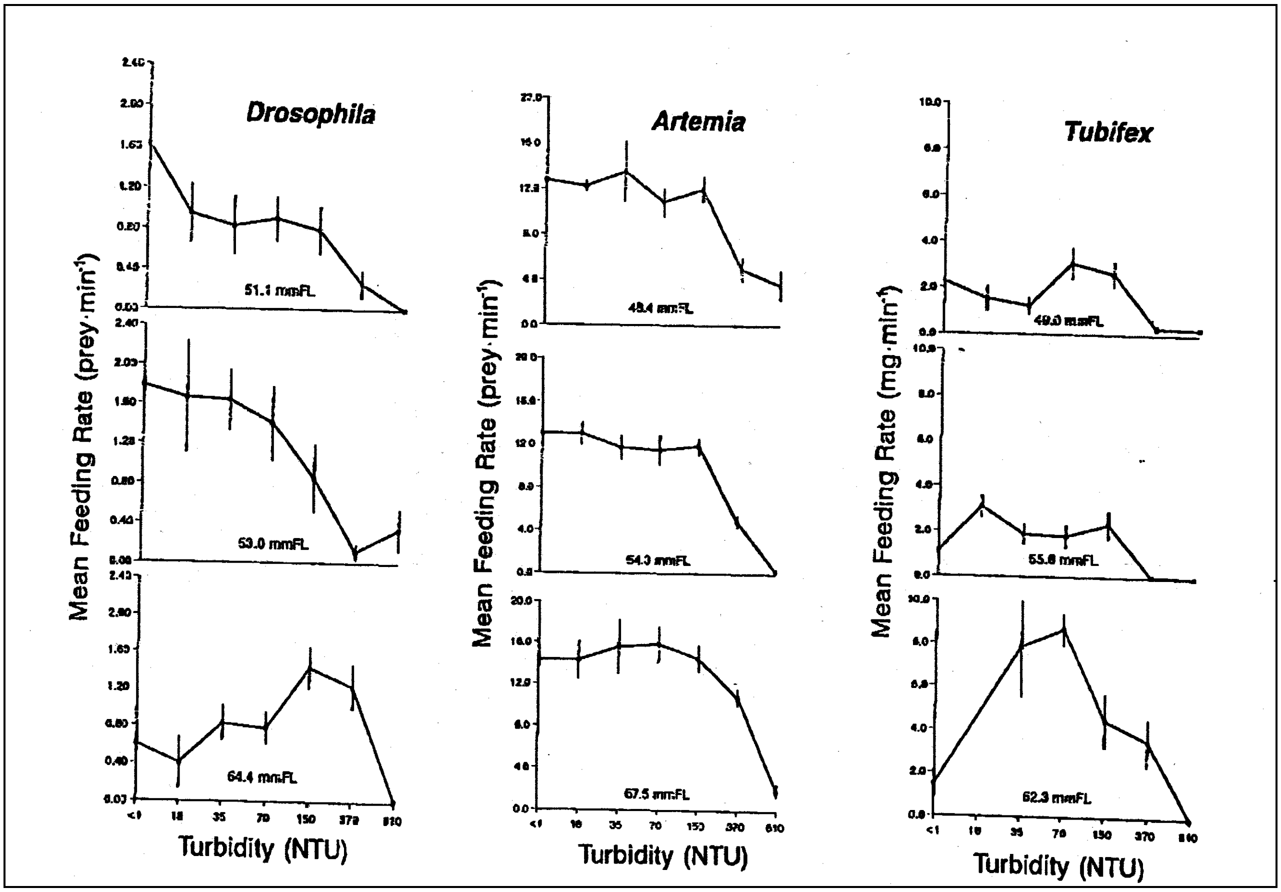

Figure 24. Feeding rates of juvenile Chinook salmon on surface, planktonic, and benthic prey at different turbidity levels. Figure 2 in Gregory (1994).

While these first two studies do indicate reduced growth for fish from exposure to turbidity, the potential of these studies to be ecologically meaningful has been reduced by field studies showing feeding success of salmonids in turbid waters (e.g., White and Harvey 2007).

One study examined both the effect of turbidity on food availability (invertebrate abundance and biomass) and the subsequent effect on fish growth. As described in the above section of turbidity effects on macroinvertebrates, Shaw and Richardson (2001) found that 23 NTU sediment pulses decreased benthic macroinvertebrate abundance and family richness, increased drift abundance, and decreased drift family richness. In a subsequent part of the experiment, they found that sediment pulses released every other day into an experimental stream for various durations reduced rainbow trout length and weight increase (Figure 27). Length gain was significantly reduced compared to control when turbidity pulses lasted a minimum of four to five hours; weight gain was reduced when turbidity pulses lasted, at a minimum, five to six hours. The authors concluded that direct effects of turbidity (reduced prey capture success and physiological stress) were more important to explaining weight reductions than food availability.

Population Studies

A few studies have examined how different turbidity levels over a long time can affect abundance, reproduction, biomass, and other measures of fish population health. However, such studies utilize population-based models based on results of other empirical studies measuring reactive distance, feeding success, and growth. For example, Harvey and Railsback (2004) modeled how turbidity affected fish abundance (coastal cutthroat trout) and biomass in twelve-year simulations. Turbidity was modeled based on data from two creeks in northwestern California and was assumed to reduce reactive distance, while simultaneously reducing the risk of predation (see below). Model results indicated that simulations with high turbidity regimes consistently produced lower levels of abundance, reproduction, and biomass than the river with a low turbidity regime, except in instances where food availability doubled. Further elaborating on the modeling approach, Harvey and Railsback (2009) examined how different patterns in food availability and foraging strategy influenced population-level outcomes in 15 year simulations of different turbidity regimes. In the simulations, the low-turbidity regime produced modest variation in biomass over time regardless of the relative importance of drift food to the simulated fish. However, fish abundance under the high-turbidity regime fell to zero unless the fish almost entirely depended on search-based strategy unaffected by turbidity. The preceding studies assumed that turbidity can affect the ability of fish to detect and capture prey. However, the authors have noted that, despite the fact that modeling would indicate reduced fish populations, there are robust populations of coastal cutthroat trout in the moderately turbid stream that was studied. Thus, there is a lack of correspondence between field observations and simulations suggesting rapid local extinction, indicating uncertainty with regard to relationships between turbidity and food supply.

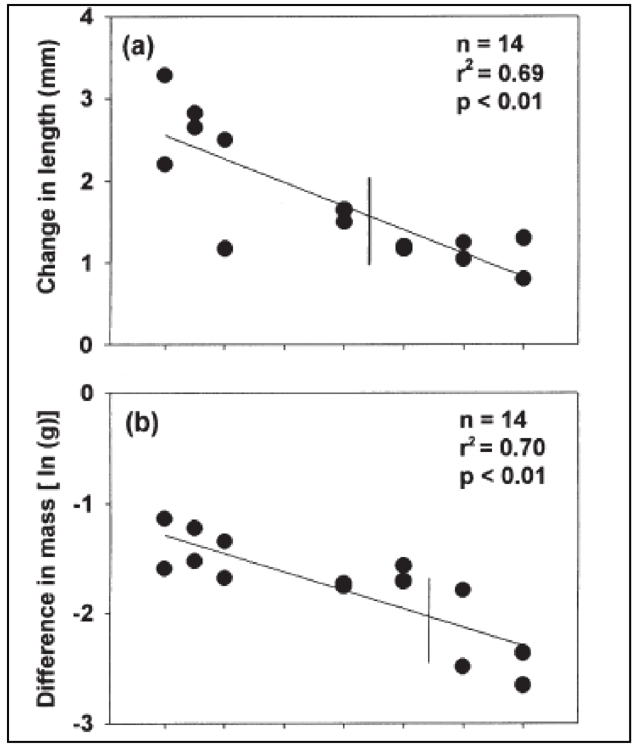

Figure 25. Sediment pulse (23 NTU) duration vs. length and mass increase in rainbow trout over 19 days. X-axis shows the number of hours fish were exposed to turbid waters every other day. From Figure 5 in Shaw and Richardson (2001).

Use of turbidity by fish as cover

Several papers indicate that juvenile fish use turbid waters as cover from predators. Gregory (1993) found that juvenile Chinook salmon exhibited a startled response to models of predators in clear water conditions, but not in turbid (23 NTU) conditions. Abrahams and Kattenfeld (1997) found that fathead minnows use "dangerous habitats" (those in proximity to a predator) more often in turbid (13 NTU) water than clear water, indicating that perceived risk is less in turbid waters. However, turbidity did not affect mortality rates of minnows in a parallel experiment conducted at 11 NTU over three days. In a study in British Columbia, Gregory and Levings (1998) found significantly less predation on juvenile Chinook salmon by piscivorous fish in the highly turbid Fraser River (27 to 108 NTUs) than in the clear water Harrison River (<1 NTU) and slightly turbid Nicomen Slough (1-6 NTUs).

Consideration of literature regarding use of cloudy water by juvenile fish suggests that, while there is a temporary benefit to increased turbidity, this benefit may be outweighed by other factors, such as effect of turbidity on growth rates of fish. Moreover, streamside vegetation (i.e., shade) appears to be more important for use as cover than turbid waters. For example, Gadomski and Parsley (2005) found decreased predation of white sturgeon with increasing turbidity; at the same time, vegetative cover also was associated with decreased predation. This is consistent with the findings of Gregory and Levings (1996) who found that streamside vegetation was more important than turbidity in providing cover for juvenile salmonids. In simulations, Harvey and Railsback (2004; 2009) included a turbidity benefit in the form of lessened predation risk and in which fish take both food acquisition and predation risk into account in habitat selection. In such cases, predators still took about the same proportion of fish because fish occupied riskier habitats under high-turbidity regimes because of reduced food intake.

The preference of juvenile fish for turbid water isn't universal. For example, Gradall and Swenson (1982) found that brook trout showed no preference for moderately (5.8 NTU) or highly turbid (56 NTU) water.

Meta-analysis of turbidity effects on fish

A difficulty in developing water quality criteria for turbidity based on the studies presented above is that duration of exposure is very different from study-to-study and thus such studies are difficult to compare to each other. Basing water quality criteria on long-term studies at high turbidity levels would overstate effects in waters that only experience short-term turbidity spikes. Conversely, water quality criteria based on short-term studies could understate chronic effects. As a way to incorporate turbidity levels *and* exposure duration into effects analysis, Newcombe (2003) developed a meta-analysis that assigned a severity of ill effect (SEV) score to the results of laboratory and field experiments. Turbidity effects considered for the model include fish reactive distance, predator prey dynamics, egg and larval development growth rates, and habitat alteration effects. Newcombe (2003) assigned SEV scores to the results of the studies, and then regressed against water clarity measurements and exposure duration from literature to develop a log-linear regression.

The Newcombe model is useful in that it provides a method to estimate the potential risk of impairment over a range of turbidity conditions and durations of potential exposure effects to clear water fish. However, the IMST (2006) questioned reliance on it to develop water quality criteria for turbidity and noted that it is not definitive in its conclusions and that it wasn't clear if the model had been validated. Smedley, et al. (2011) found poor correlations between the SEV index and predicted populations of brook trout and slimy sculpin in New Brunswick. The model was calibrated based on studies reporting visual clarity as a black disk sighting range and beam attenuation (see Chapter 2 for a discussion of these measures of clarity). The relationship between turbidity and other measures of clarity is site-specific. As such, caution should be exercised when extrapolating these results to Oregon or other geographic locations. Another caution in using Newcombe's model is that it was developed from turbidity-effects literature reported from around the world and thus is not necessarily specific to Oregon species.

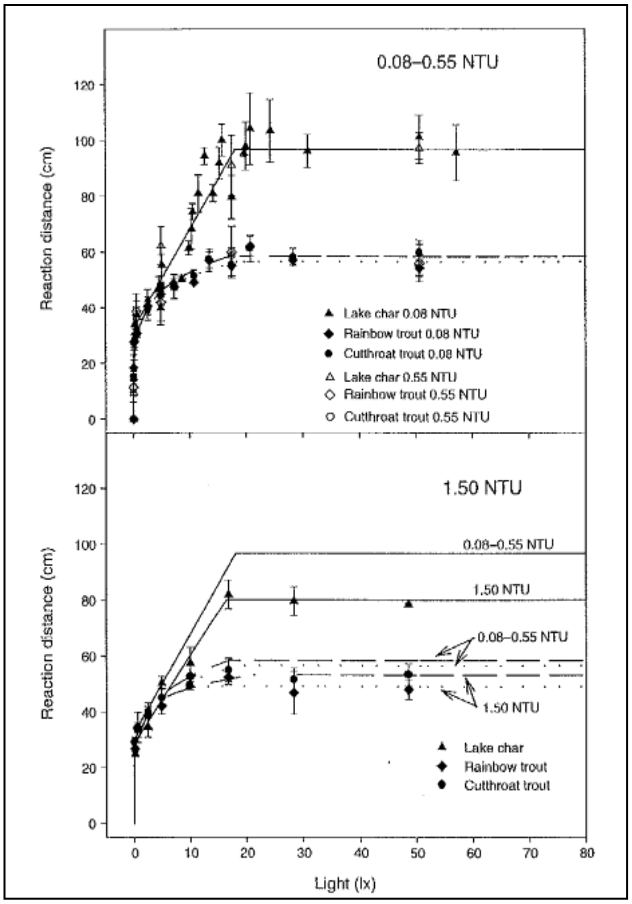

Figure 26. Relationship between turbidity and reactive distance of trout at different light intensities (Figure 1 in Mazur and Beauchamp 2003).

Literature regarding effects of turbidity on fish - lakes

Summary

In general, effects of turbidity on lake fish are similar to those in fish found in streams. However, research focuses primarily on reactive distance studies and feeding rates. Reactive distance studies appear to focus on the minimum turbidity at which reactive distance appears to be affected; these studies appear to coalesce at turbidities of 1-3 NTU. Feeding rate appear to be affected in some fish, such as bass, Lahontan redside shiner and Lahontan cutthroat trout at levels as low as 5-10 NTU; however, one study found that feeding rate of largemouth bass was not affected at 37 NTU. Bluegill feeding rate was affected at 60 NTU; sediment-tolerant fish such as crappie are generally not affected by high turbidity.

DEQ was unable to find any studies examining how turbidity may affect growth rate in lake fish found in Oregon and only a few indirect or anecdotal studies discussing effects of turbidity on overall populations of fish; none of the latter were framed in a way to elaborate to a numeric water quality standard.

Reactive Distance Studies

Hansen, et al. (2013) found that the reactive distance of yearling Chinook salmon began to decrease exponentially beginning at turbidities of 1.65 NTU. Vogel and Beauchamp (1999) determined that reactive distances in lake trout being fed juvenile rainbow trout decreased at 3.18 and 7.40 NTUs but not at the lowest turbidity level tested (0.9 NTUs) as compared to reactive distance in clear water. Mazur and Beauchamp (2003) found that reactive distance of lake, rainbow, and cutthroat trout didn't decrease when turbidity increased from 0.08 to 0.55 NTU, but did decrease at 1.50 NTU (Figure 28), suggesting a threshold turbidity exists between those levels consistent with the findings of Vogel and Beauchamp (1999). Crowl (1989) found the reactive distance of largemouth bass to be significantly less in turbid (~18 JTU) water than in clear water. Miner and Stein (1996) found in a laboratory experiment that reactive distance of bluegill decreased as a power function of turbidity with a 50% reduction occurring at 1.2 NTU.

Feeding rate studies

Feeding rates of Lahontan redside shiner and cutthroat trout on daphnia decreased 60-80% when turbidity increased from from 3.5 NTU to 25 NTU (Figure 29; Vinyard and Yuan 1996). Decreases in feeding rate were evident as low as 6-10 NTU, although the differences at these levels were not tested for significance. Carter, et al. (2010) found that prey consumption of smallmouth bass decreased as turbidity increased from 0-40 NTU, with significant decreases in consumption in the lowest turbidity level tested (5 NTU). Similarly, Shoup and Wahl (2009) found that size selectivity of prey by largemouth bass was impacted at 10 NTU (19-49 hour trials) and 40 NTU (42-77 hours), and a reduced overall predation rate at 40 NTU compared to 0 and 5 NTU treatments. Reid, et al. (1999) in one hour trials found that predation rates of largemouth bass were affected at 70 NTU compared to a clear water control, but not at 18 or 37 NTU. Gardner (1981) found that prey consumption rates decreased in bluegill at 60, 90, and 120 NTU compared to a clear water control.

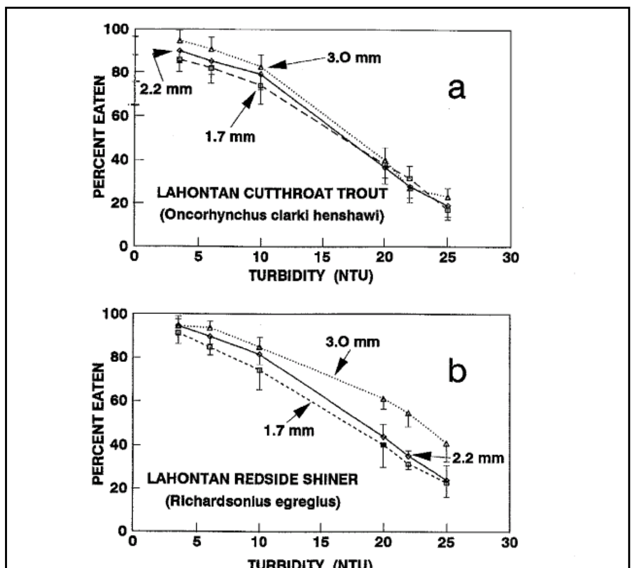

Fig. 1. Mean percent prey consumed in relation to turbidity. Upper panel (a) shows results from feeding trials with Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*), and lower panel (b) shows results from Lahontan redside shiner (*Richardsonius egregius*). Four fish of each species were exposed to prey of a single size for 2-h feeding trials. *Daphnia magna* prey sizes are as indicated. Vertical bars indicate 1 standard deviation.

Figure 27. Relationship between turbidity and feeding rate of Lahontan cutthroat trout and redside shiner. Figure 1 in Vinyard and Yuan (1996).

A few studies indicate that turbidity does not affect certain fish. Rowe, et al. (2003) found that the feeding rates of rainbow trout from New Zealand lakes did not decrease at 160 NTU over controls.

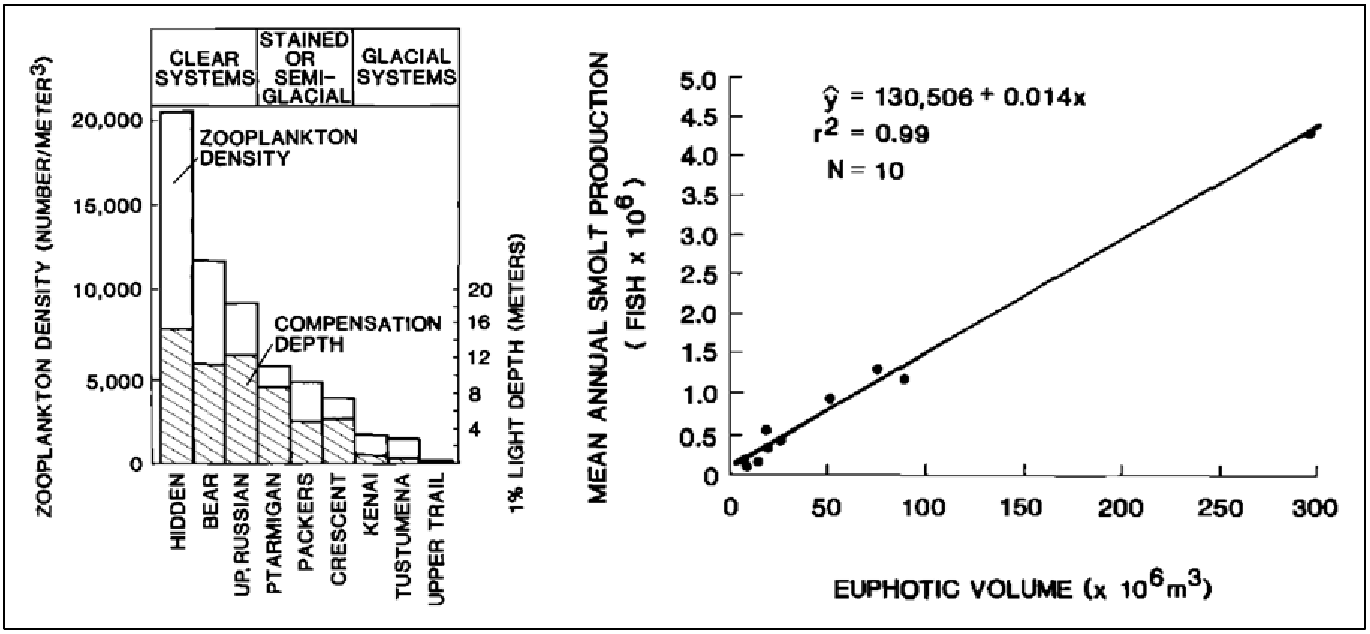

Figure 28. Zooplankton density in clear, semi-glacial, and glacial lakes, and relationship of sockeye smolt production to euphotic volume. Figures 3 and 4 in Lloyd, et al. (1987).

However, the study found that in clear water, rainbow trout ate primarily larger prey, whereas, selectivity decreased as turbidity increased. The study did not report if the change in size selectivity affected growth rates. In another study, growth rates of juvenile white and black crappie were not affected by turbidities ranging from 7 to 174 FTU and growth rates of adult crappie were not affected in 13-144 FTU treatments in 25 week studies (Spier and Heidinger 2002). Crappie are generally thought to be tolerant to changes in turbidity and other measures of water quality (Buck 1956).

Growth and population studies

DEQ was unable to find any studies examining how turbidity may affect growth rates in lake fish. However, in one study, Lloyd, et al. (1987) noted that the number of juvenile sockeye salmon in glacial lakes in Alaska, measured as number of outmigrating smolts, related significantly (r^2=0.99) to euphotic volume, which decreases due to increases in turbidity; however, information was not available to indicate a level of turbidity that might be associated with a specific decrease in number or density of smolts (Figure 30). Shrader (1999) predicted that a 10% decrease in suspended solids in the Prineville Reservoir in Oregon would lead to an approximate 17% increase in fish yield due to increased primary production. Most other studies examining fish abundance and turbidity are anecdotal. For example, Ewing (1991) found that chronic turbidity levels greater than 100 formazin turbidity units (FTU) were the likely causal factor for the small fish population in a Louisiana bottomwood backwater system. When turbidity levels decreased as a result of restoration of natural flood patterns, fish populations of centrarchids, such as sunfish and bass, increased markedly. Buck (1956) found a much greater total weight of fish per acre in clear (average turbidity 25 ppm) ponds than in intermediate (25-100 ppm) or high-turbidity (>100 ppm) ponds in Oklahoma.[7] He noted that bluegills and redear sunfish were particularly affected. On the other hand, Bachmann, et al. (1996) found a slightly negative (r^2 = 0.17) correlation between Secchi depth transparency and standing crop (kg/ha) of fish in 65 Florida lakes.

[7] In this case, turbidity most likely refers to total suspended sediment or suspended sediment concentration.

Literature Regarding Effects of Turbidity on Fish – Estuaries and Marine

Studies examining the effects of turbidity on estuarine fish indicate that feeding is optimal at moderate turbidity levels as compared to clear water or highly turbid conditions. Boehlert and Morgan (1985) found that juvenile Pacific herring feed optimally at suspended sediment concentrations of 500-1000 mg/L, but exhibited less feeding in clear water and in sediment concentrations higher than 1000 mg/L. Gregory (1990), examining foraging behavior of juvenile Chinook salmon in estuarine conditions, found that, while reactive distance declined with turbidity, feeding rates on benthic Tubifex were highest at 50-100 mg/L suspended sediment, and were less in clear water and in concentrations higher than 100 mg/L (Figure 31). Quesenberry, et al. (2007) found that, although turbidities of 40-80 NTU affected reactive distance of threespined stickleback, it had no effect on foraging success. Gregory (1990) suggests that reduced risk from predation may allow for more foraging.

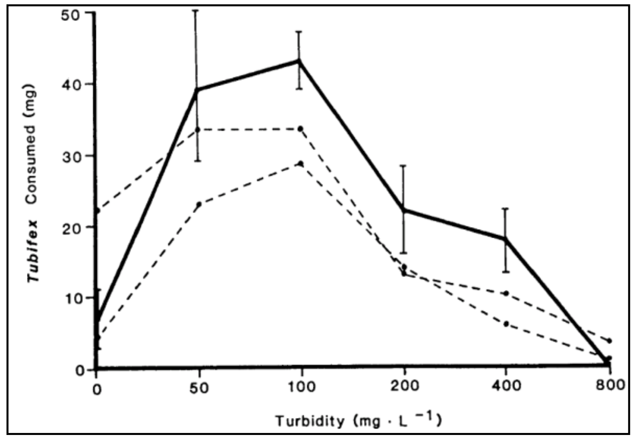

Figure 29. Relationship between turbidity and feeding rates of juvenile Chinook salmon on *Tubifex*. Figure 4 in Gregory (1990).

Sources of uncertainty

As noted at the beginning, studies examining the impact of turbidity on prey detection (reactive distance) and feeding in fish provide a relevant base for understanding how turbidity may affect aquatic life. However, such studies generally measure effects at turbidity levels that frequently occur naturally and which likely do not affect fish populations. Unfortunately, longer-term studies examining the effect of turbidity on populations utilize models and may not capture all relevant variables (Harvey and Railsback 2009). Studies measuring fish growth are therefore likely to be the most relevant to developing a water quality standard in Oregon. Only three studies were done of fish present in Oregon waters (Sigler, et al. 1984; Shaw and Richardson 2001; and Sweka and Hartman 2001b). The Sigler study found significant effects at the lowest level of turbidity measured (22-23 NTU) for 11-15 days. Thus, it is not clear if lower levels of turbidity may also affect growth, whether at the same duration or in shorter or longer durations. Thus, there is still some uncertainty with respect to a lowest effects threshold; likely the best way to overcome this uncertainty is to develop site-specific or ecoregion specific relationships between fish growth and turbidity (B. Harvey, pers. comm., 12/13/13). In the absence of such information, DEQ may need to rely on more conservative assumptions in order to protect aquatic life.

In lakes, as in streams, research on the effects of reduced water clarity on fish focuses primarily on changes in predator-prey dynamics due to changes in reactive distance of predatory fish, in particular species of centrarchidae (sunfish). Results from these studies indicate feeding effects at turbidity levels as low as 5 NTU for relatively short (3.5-42.6 hour) exposures. However, studies have found that certain fish in lakes and reservoirs, such as crappie, appear to be more tolerant of turbid conditions. There is a lack of studies examining how turbidity may affect growth rate and population dynamics in fish sensitive to turbidity. A few studies focus on chronic turbidity effects on fish density, presumably due to decreased food availability; however, these studies are generally anecdotal and are not useful for setting a water quality standard for turbidity.

Summary tables of aquatic life effects in streams and lakes.

Tables 4 and 5 summarize studies presented in this section that could be used to equate a specific turbidity level to potential effects on aquatic life in streams and lakes, respectively. The tables include relevant duration of exposure data where available. One of the major uncertainties in presenting such a table is that studies utilize different instruments to measure turbidity and some do not describe their instrumentation at all. Thus, comparison of studies is subject to some uncertainty.

Table 4. Summary of effects of turbidity on aquatic life in streams.

Turbidity Level (margin of error)	Duration	Effect	Source	Turbidity Measurement	Type of Study
Effects at reported turbidity levels at ≤10 turbidity units					
4-8 NTU	n/a (reference site approach)	Decrease in *Epeorus* species in Umatilla River	Scherr, et al. (2011)	LaMotte 2020	Field
4.4 NTU	n/a (reference site approach)	85% chance of stream being impacted (EPT index <18)	Paul (unpub.)	Various	Field
5 NTU	none given	Modelled decrease in primary productivity in clear Alaska streams by 3-13% (stream depth 0.1 – 0.5 m)	Lloyd, et al. 1987	Hach "Portalab"	Field
7 NTU	Two months	75% decrease in benthic algal biomass	Davies-Colley, et al. 1992	Hach 2100A	Field
7 NTU	Two months	70% decrease in macroinvertebrate density	Quinn, et al. 1992	Hach 2100A	Field
7-25 NTU	n/a	Decrease in macroinvertebrate density and other measures of macroinvertebrate health	Prussian, et al. 1999		
9 NTU	n/a	20% decrease in PREDATOR score using Oregon data	ODEQ turbidity data	n/a	Field
10 NTU	15 minutes	50% decrease in brook trout reactive distance	Sweka and Hartman 2001a	Lamotte 2020 turbidimeter	Laboratory
10 NTU	5 days	20% decrease in brook trout growth	Sweka and Hartman 2001b	Lamotte 2020 turbidimeter	Laboratory
10-60 NTU	4-6 days	Decrease in prey consumption by juvenile coho salmon after initial exposure to 60 NTU; also, higher response time and increased number of mis-strikes at prey.	Berg 1982	DRT-150 Turbidimeter	Laboratory
Effects at reported turbidity levels from 11-20 turbidity units					
11-32 NTU	14 days	Reduced weight and length gains in newly emerged coho salmon (raceway channels)	Sigler, et al. 1984	Hach 2100A Turbidimeter	Laboratory
15 NTU	n/a	20% reduction in rainbow trout reactive distance	Barrett, et al. 1992	Not reported	Laboratory (artificial stream channel)
18 NTU	1-10 minutes	Reduced feeding rates of small-medium juvenile Chinook salmon on surface prey	Gregory 1994	Fisher DRT-400 Turbidimeter	Laboratory
20 NTU	One hour	Reduced prey capture success by juvenile coho salmon	Berg and Northcote 1985	Fisher 400 DRT Turbidimeter	Laboratory

Turbidity Level (margin of error)	Duration	Effect	Source	Turbidity Measurement	Type of Study
Effects at turbidity levels from 21-30 turbidity units					
22 NTU	11 days	Reduced weight and length gains in newly emerged coho salmon (oval channels)	Sigler, et al. 1984	Hach 2100A Turbidimeter	Laboratory
23 NTU	1-6 hour daily pulses over 9 and 19 days	Reduced abundance and species richness of benthic macroinvertebrates. In addition, reduced rainbow trout length and weight gain when turbidity pulses lasted 4-5 and 5-6 hours, respectively.	Shaw and Richardson 2001	Not reported (converted from suspended sediment concentrations, but does not report relationship)	Laboratory
23 NTU	12 days	Reduced startle response by juvenile Chinook salmon	Gregory 1993	Fisher DRT-400 Turbidimeter	Laboratory
25 NTU	none given	Modelled decrease in primary productivity in clear Alaska streams by 13-50% (stream depth 0.1 – 0.5 m)	Lloyd, et al. 1987	Based on information using Hach "Portalab"	
25 NTU	15 minute	Reduced drift prey foraging success	Harvey and White 2008	DTS-12	Laboratory
25-35 NTU	3 months	Decrease in whole stream metabolism	Parkhill and Gulliver 2002	Not reported	Controlled field (laboratory streams)
27+ NTU	1.5 hours	Predation rates on juvenile Chinook salmon by piscivorous fish significantly reduced in the Fraser River	Gregory and Levings 1998	Fisher DRT-100 Turbidimeter	Field
30 NTU	n/a	55% reduction in rainbow trout reactive distance	Barrett, et al. 1992	Not reported	Laboratory (artificial stream channel)
30 NTU	One hour	Decrease in reactive distance, capture success and percentage of prey ingested for juvenile coho salmon. In addition, dominance hierarchies broke down and gill flaring occurred more frequently	Berg and Northcote 1985	Fisher 400 DRT Turbidimeter	Laboratory
30 NTU	24 hours	Increased cough frequencies in coho salmon	Servizi and Martens 1992	HF Instruments DRT 100	Laboratory
Effects at turbidity levels from 31-50 turbidity units					
38 NTU	19 days	Decreased weight and length gains of newly emerged steelhead (raceway channel)	Sigler, et al. 1984	Hach 2100A Turbidimeter	Laboratory

Turbidity Level (margin of error)	Duration	Effect	Source	Turbidity Measurement	Type of Study
42 NTU	96 hours	25% increase in blood sugar levels in coho salmon	Servizi and Martens 1992	HF Instruments DRT 100	Laboratory
45 NTU	19 days	Decreased weight and length gains of newly emerged steelhead (oval channel)	Sigler, et al. 1984	Hach 2100A Turbidimeter	Laboratory
50 NTU	5 days	50% decrease in brook trout growth rate	Sweka and Hartman 2001b	Lamotte 2020 Turbidimeter	Laboratory
50 NTU	15 minutes	Decrease in proportion of drift prey consumed in juvenile cutthroat trout and coho salmon	Harvey and White 2008	DTS-12	Laboratory
50 NTU	15 minutes	Decrease in proportion of live oligochaetes drifting along an experimental stream bottom by juvenile cutthroat trout	Harvey and White 2008	DTS-12	Laboratory
Effects at turbidity levels >50 turbidity units					
60 NTU	One hour	66% reduction in juvenile coho salmon reactive distance (did not return to normal levels after pulse decreased)	Berg and Northcote 1985	Fisher 400 DRT Turbidimeter	Laboratory
70 NTU	30 minutes	Avoidance of juvenile coho salmon to turbid waters	Bisson and Bilby 1982	Not reported	Laboratory
80 NTU	96 hours	50% increase in blood sugar level in coho salmon	Servizi and Martens 1992	HF Instruments DRT 100	Laboratory
150 NTU	15 minutes	Decrease in proportion of benthic prey consumed by juvenile cutthroat trout and coho salmon	Harvey and White 2008	DTS-12	Laboratory
170 NTU	Ten days	50% decrease in productivity and 60% decrease in chlorophyll *a* concentrations	Van Nieuwenhuyse and LaPerreriere (1986)	Hach Portolab	Laboratory

Table 5. Summary of effects of turbidity on aquatic life in lakes and reservoirs.

Turbidity Level	Duration	Effect	Source	Turbidity Measurement	Lab or Field
Effects at turbidity levels ≤10 turbidity units					
~1.2 JTU	chronic	50% decrease in reactive distance of bluegill trout to avoid largemouth bass	Miner and Stein 1996	Not reported	Laboratory
1.5 NTU	4 hours	Minimum turbidity to decrease reactive distance of lake, rainbow, and cutthroat trout	Mazur and Beauchamp 2003	LaMotte 2008	Laboratory
1.65 NTU	1 hour	Lowest effect level for turbidity to decrease reactive distance in yearling Chinook salmon	Hansen, et al. (2013)	LaMotte 2020e	Laboratory
3.18 NTU	4 hours	Decrease in reactive distance of lake trout to juvenile rainbow and cutthroat trout at optimum light intensity	Vogel and Beauchamp 1999	LaMotte 2008	Laboratory
5 NTU	n/a	80% reduction in compensation depth	Lloyd, et al. 1987	HF DRT-150 Turbidimeter	Field
5 NTU	3.5 – 42.6 hours	Significant decrease in consumption of prey by smallmouth bass	Carter, et al. 2010	LaMotte 2020	Laboratory
10 NTU	19-49 hour	Change in size selectivity of prey by largemouth bass	Shoup and Wahl 2009	Cole-Parmer Model 8391–40	Laboratory
Effects at turbidity levels from 11-20 turbidity units					
17-19 JTU	n/a	Decrease in reactive distance of largemouth bass to crayfish	Crowl 1989	Not reported (Jackson turbidimeter)	Laboratory
Effects at turbidity levels from 21-30 turbidity units					
25 NTU	2 hours	60-80% decrease in feeding rates of Lahontan redside shiner and cutthroat trout on daphnia	Vinyard and Yuan 1996	DRT-15 Turbidimeter	Laboratory
Effects at turbidity levels from 31-50 turbidity units					
30+ NTU	n/a	Limitation in compensation of photosynthetic efficiency for low-light conditions	Lloyd, et al. 1987	n/a	Field
33 NTU	n/a (mean turbidity over multiple	Reduction in chlorophyll *a* levels in glacial lakes	Koenings, et al. 1990	DRT-100	Field

Turbidity Level	Duration	Effect	Source	Turbidity Measurement	Lab or Field
	lakes and years)				
40 NTU	42-77 hours	Decrease in predation rate by largemouth bass	Shoup and Wahl 2009	Cole-Parmer Model 8391–40	Laboratory
Effects at turbidity levels >50 turbidity units					
60 NTU	3 minutes	Decrease in prey consumption by bluegill	Gardner 1981	DRT-100	Laboratory
70 NTU	one hour	Decrease in predation rates by largemouth bass	Reid, et al. 1999	DRT-15B	Laboratory
100 FTU	n/a	Population level declines of centrarchids in a Louisiana bottomwood backwater system	Ewing 1991	Hach DR-EL/1	Field
144 FTU	25 weeks	No effect on growth rate of adult crappie	Spier and Heidinger 2002	Hach DR-2000	Field
160 NTU	3 hours	No decrease in predation rate by rainbow trout; however, size selectivity was affected.	Rowe, et al. 2003	Hach 18910 Turbidimeter	Laboratory
174 FTU	25 weeks	No decrease in growth rates of juvenile white and black crappie	Spier and Heidinger 2002	Hach DR-2000	Field

Chapter 4. Effects of Turbidity on Recreation and Aesthetics

Increased turbidity levels can affect recreational use of waters in Oregon and elsewhere, both directly and indirectly. Directly, turbidity reduces visibility, which can diminish "suitability" of waters for swimming (Smith, et al. 1991) and fishing (Lloyd, et al. 1987). Indirectly, turbidity induced reductions in fish populations can reduce catch rates and reduce "desirable species" (Buck 1956).

Effects of turbidity on aesthetics and swimming

Turbidity can have deleterious effects on perceptions of water quality, which may in turn reduce use of waters for recreational uses and swimming. Most of the research that DEQ found related to the effects of turbidity and visual clarity on perceptions of aesthetics and the "suitability" of waters for use has been conducted in New Zealand. Smith, et al. (1991) in a New Zealand survey found that 60% of people considered waters yielding a black disk sighting range of 1.2 meters (approximate turbidity 4 NTU) to be suitable for swimming. In the same study, 90% of those surveyed considered waters yielding a black disk sighting range of 2.2 meters (~1-2 NTU) suitable for swimming.[8] In another study, Smith and Davies-Colley (1992) surveyed 15 field staff of the New Zealand Water Resource Survey in New Zealand with respect to recreational bathing and aesthetic suitability-for-use in streams. Results are summarized in Table 6.

Table 6. Relationship between turbidity and suitability of water for swimming and overall aesthetics. Data from Smith and Davies-Colley, 1992.

	Swimming Suitability (NTU)	Aesthetic Suitability (NTU)
Eminently suitable	≤1	≤1
Suitable	>1 - 2	>1 - 2
Marginally suitable	>2 - 3	>2 - 3
Unsuitable	>3 - 8	>3 - 11
Totally unsuitable	> 8	>11

In laboratory tests, Smith and Perrone (1996) observed that the percent change in clarity required to present perceptible differences to surveyed viewers decreased as the control condition for comparison increased in turbidity. A greater than 300% (or 15 NTU) increase above a turbidity sample of 5 NTUs was needed to reveal a 'conspicuous' difference between samples; with a similar response by those surveyed at approximately 16 NTUs (160%) above a control sample of 10 NTUs, and 70 NTUs (140%) above a control sample of 50 NTUs. 'Somewhat of a noticeable difference' was perceived at 8.5 NTUs (170%), 9 NTUs (90%), and 35 NTUs (70%) above 5, 10, and 50 NTU control levels, respectively. 'Barely noticeable differences' occurred at 3.4 NTUs (68%), 3.2 NTUs (32%), and 10 NTUs (20%) above 5, 10, and 50 NTU control levels, respectively. The appearance of disparity between these statistics and Table 6 results may be due to perceived differences above perfectly clear water (0 NTUs) and perceived differences above turbidity levels of 5 NTUs or greater.

One area of uncertainty in presenting these studies is the applicability of New Zealand studies to impacts on recreation in Oregon. However, there are a number of similarities between Oregon and New Zealand. For example, Oregon (especially Western Oregon) and New Zealand have a comparable array of lakes,

[8] Equivalent turbidity calculations here and in Table 4 based on equations in Smith, et al. (1997)

reservoirs and streams. New Zealand has many highly oligotrophic and transparent lakes as well as many eutrophic and dystrophic lakes which are not very transparent. Streams and rivers include a wide variety of turbidity regimes as well (R. Petersen, *pers. comm.*). Smith, et al. (1995 a, b) noted that the perception of a water body depends on the use an observer expects to make of it as well as the observer's prior experience. For example, if the water quality was perceived to be "natural", users were more accepting of reduced transparency or color resulting from dissolved organic matter. In as much as people in both Oregon and New Zealand have an equivalent "reference set" of natural water bodies on which to base their opinions, the results of research in New Zealand are applicable as a basis for setting criteria in Oregon.

Effects of turbidity on fishing

In locations where chronic turbidity results in decreased fish populations and diversity, a number of studies have noted an indirect effect on the quality of fishing in those locations. For example, Buck (1956), in a study of a clear and a turbid reservoir in Oklahoma, found that fish species grew faster in the clear reservoir. In addition, catch per unit effort in the clear reservoir was reported as 3-4 times higher in the clear reservoir than the turbid reservoir. Drenner, et al. (1997) found that catch rates of largemouth bass were significantly and linearly correlated with turbidity in an experimental pond (Figure 32). Ewing (1991) hypothesized that chronic turbidity (>100 NTU) was the culprit for the decline in fish populations in a bottomland hardwood backwater system. Lloyd, et al. (1987) reported a 55% decline in sport fishing downstream from mine discharges on the Chatainika River, Alaska, which was attributed to avoidance by fishers of increased turbidities of 8-50 NTU. The authors did not note whether this decline was due to a decrease in fish numbers or a preference to fish in clear waters due to safety or aesthetic concerns.

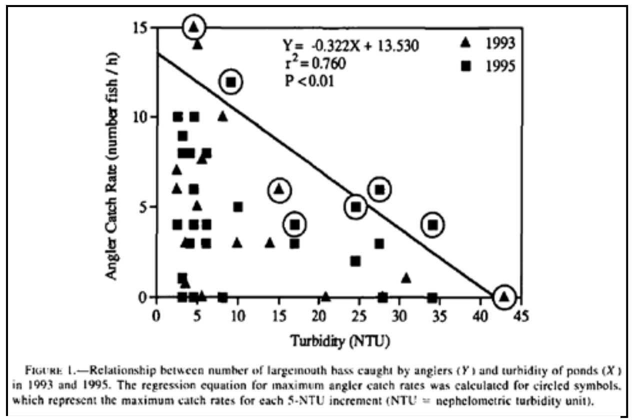

Figure 30. Relationship between turbidity and angler catch rate of largemouth bass in an experimental pond (Drenner, et al 1997).

Chapter 5. Effects of Turbidity on Drinking Water Treatment

Analyses conducted by DEQ and others has found that high turbidity levels in drinking water sources may either prevent drinking water treatment operators from providing safe drinking water to their communities or make treatment of drinking water more expenstive. Because domestic water supply is a beneficial use under the Clean Water Act, DEQ has compiled relevant data in order to review the potential effects of turbidity in the context of whether DEQ should set a water quality standard to protect domestic water supplies.

The Safe Drinking Water Act (SDWA) regulates the level of turbidity in drinking water,[9] as higher turbidity levels are often associated with higher levels of disease-causing microorganisms such as viruses, parasites and some bacteria. Most drinking water treatment facilities run by public water systems (PWSs) have the capacity to remove turbidity-causing sediments during treatment of raw water; however, the amount of turbidity that can be effectively removed depends on the treatment technology in use (US EPA 1999). Systems using slow sand filtration or similar technologies will usually need to shut down when turbidities exceed 5 NTU (National Drinking Water Clearinghouse 1996). Some systems in Oregon with frequent high turbidity install advanced filtration systems that can treat water with turbidity higher than 50 or 100 NTU, but these systems are expensive and may not be affordable for all small PWSs (ODEQ 2010*)*. Moreover, more advanced treatment systems must use higher levels of flocculent and coagulant as source water turbidity increases, at a higher cost to ratepayers.

DEQ conducted a study of turbidity for public water systems (PWSs) in Oregon with drinking water source areas (DWSAs) in the North and Mid Coast Ranges (ODEQ 2010). The report includes case studies of eight PWSs and status reports of an additional ten PWSs. Some systems, such as Falls City and the Arch Cape Water District, must cease purification at turbidities higher than 5 NTU to prevent their filtration systems from clogging. Such systems have approximately 2-3 days of storage capacity. Other systems, such as Astoria and Forest Grove, may switch from a primary to a secondary DWSA in the case of high turbidity. The City of Astoria does not have storage capacity, meaning that they must jockey between various sources to ensure that the public has access to drinking water if turbidity is high (Evan Hofeld, Oregon Health Authority, *pers. comm.*) The City of Yamhill's system can handle episodes of high turbidity, but maintenance and treatment costs increase during these episodes.

For systems using flocculation and sedimentation, studies have linked turbidity to higher drinking water treatment costs. Moore and McCarl (1987) studied overall costs of sediment in the Willamette Valley including those related to drinking water treatment using data from the water treatment in Corvallis. The study indicated that a 1% decrease in turbidity would reduce sediment-related treatments costs (cost of alum, lime, and associated maintenance) by 0.35%.

[9] The SDWA requires that for systems that use conventional or direct filtration, turbidity can go no higher than 1 NTU and samples for turbidity must be less than or equal to 0.3 NTUs in at least 95 percent of the samples in any month. Systems that use filtration other than the conventional or direct filtration must ensure that turbidity at no time exceeds 5 NTUs or follows more restrictive state limits, if they exist.

The findings of Moore and McCarl (1987) were similar to those of studies in other areas of the United States. Dearmont, et al. (1998) found in a study of 12 treatment plants in Texas that elasticity of cost of chemicals to treat water with respect to turbidity was 0.25.[10] Foca (2002) studied two water systems in North Carolina serving approximately 25,000 people and found that, if turbidity was fixed to an average of 5 NTU, annual savings could be $7200. Forster, et al. (1987), in a study of twelve treatment systems in Ohio, found that a 25% reduction in soil erosion statewide could result in a $2.7 million savings in water treatment costs. Holmes (1988) estimated that the cost of treating suspended sediment nationally ranged from approximately $35 million to $661 million.

[10] Elasticity of cost indicates the change in demand due to increased or decreased cost. In this case, it means that for every 1% of increase in cost for drinking water due to increased turbidity, demand for water decreases by 0.25%.

Chapter 6. Conclusions and Data Gaps

Ideally, DEQ would design and conduct studies examining how turbidity affects designated uses in Oregon streams, particularly aquatic life (C. Flinders, *pers. comm.*., 12/17/13); however, such an effort is currently beyond DEQ's ability. As a result, DEQ must rely on existing data and make a judgment on the appropriate level of protection for designated uses given the available literature and in consideration of uncertainty in the studies.

The body of literature indicates that effects on aquatic life (particularly with respect macroinvertebrate abundance and diversity) are associated with turbidity levels of 4-8 NTU. At the lower end of this range, effects are focused on species generally thought to be sensitive to suspended and bedded sediment, such as Ephemeroptera, Plecoptera, and Trichoptera invertebrates; effects have been measured more generally at 7-10 NTU. However, DEQ is unaware of any study that empirically measures primary productivity effects when turbidity is less than 8 NTU. A few studies indicate that increased photosynthetic efficiency can compensate for effects of turbidity on primary productivity; however, this may come at a cost to overall biomass and cover.

Reactive distance of fish decreases with increased turbidity levels. At the same time, many studies show that fish, particularly salmonids, are able to feed even at moderate turbidities, although overall feeding rates may decrease. Moreover, increases in turbidity may cause fish to switch to a more active foraging strategy, ultimately reducing growth without a corresponding increase in food availability. Adverse growth effects on fish have been shown at turbidities as low as 22 NTU after exposure for 2-3 weeks; however, there are no studies that have studied if there are growth effects at lower turbidities and shorter durations. Pulses of turbidity (four-to-six hour pulses) of 23 NTU released every other day have been shown to affect fish growth (Shaw and Richardson 2001).

Literature shows that juvenile fish can utilize turbidity as cover from predators. Several studies indicate preferential use of turbid over clear waters in juvenile fish. However, juvenile fish also exhibit reduced anti-predator behavior in turbid waters, resulting in no significant change in predation rate compared to clear water. Moreover, streamside vegetation has been shown to be more important than turbidity as cover in one study. Even if turbidity does provide cover, the beneficial effect could be offset by reduced foraging success.

Although there are studies generally indicating that turbidity in lakes can result in similar effects on aquatic life as in streams (decreased primary productivity, macroinvertebrate abundance, and reduced fish growth), none provide a good basis for determining the turbidity level at which such effects occur. Several studies examine how turbidity reduces fish foraging success, but none measure how turbidity affects growth of lake fish and even this response variable falls short of predicting consequences for population dynamics.

In estuaries, reduced light penetration limits growth of submerged aquatic vegetation (SAV) in some areas; however, many other factors, such as current velocity, sediment characteristics, temperature, and salinity, also may be important. The U.S. EPA derived a significant relationship between water clarity and SAV growth in the Yaquina Bay and recommended light attenuation coefficients of 0.8 m^{-1} in the marine-dominated portion of the estuary and 1.5 m^{-1} in the riparian-dominated portion of the estuary to protect SAV growth. However, additional data is needed to determine light requirements for SAV growth in other estuaries. For fish in estuaries, the evidence suggests that juvenile fish prefer moderate levels of suspended sediment to balance the need for protection against predators with effects of turbidity on foraging.

Primary data gaps with respect to turbidity effects on aquatic life include:

- *The lack of controlled studies of the effect of turbidity on macroinvertebrates and primary productivity in Oregon.*
- *The lack of multi-year studies regarding the effect of chronic high turbidity, particularly examining secondary and higher levels of production.* Such studies would provide helpful data to determine if reduced primary productivity and macroinvertebrate abundance result in reductions in fish and populations.
- *The lack of testing on the effects of turbidity on fish growth below 20 NTU and less than 2-3 weeks.* Without such information, it is difficult to determine a threshold above which fish growth effects occur.
- *The general lack of studies that would provide lowest effect thresholds regarding effects of turbidity on aquatic life in lakes.* While studies indicate that turbidity acts similarly in lakes as in rivers, no studies provide a basis for a numeric standard.
- *Studies examining effects of turbidity on SAV in Oregon estuaries other than Yaquina Bay estuary*. EPA data suggest that turbidity is only one of many factors affecting SAV. Information is not available that would support development of a statewide standard.

Studies in New Zealand indicate that turbidities as low as 2 NTU may affect public perceptions of the suitability of water for swimming and aesthetics. Perceptions of recreational users of Oregon waters are likely somewhat similar to those in New Zealand, given similarities in climate and types of lakes and rivers in the two locations. Effects of turbidity on fishing are primarily related to catch rates; thus, setting a water quality standard for turbidity based on aquatic life protection should simultaneously ensure that fishing is protected.

Studies have shown that turbidity reductions from source waters can reduce treatment costs for public water systems (PWSs). However, such costs appear to be minor when spread across all users of a system. Frequent occurrences of turbidity levels 5 NTU or higher, or an occurrence lasting several days, can cause PWSs using slow sand filtration and other conventional filtration systems to shut down preventing Oregon populations from having access to municipal drinking water.

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Appendix A. Sources of turbidity in Oregon

Natural Sources of Turbidity

Natural weathering and decomposition of rocks, soils, and dead plant materials and the transport or dissolution of the weathered products in water contributes a natural "background" of turbidity-causing suspended and dissolved materials to natural waters (Sorensen, et al. 1977). Transport of sediment and organic matter in mountain streams is the result of numerous interacting processes (Beschta, et al. 1981). Vegetation absence or loss from natural attrition, windthrow, fire, and/or seismic events, along with precipitation (or wind) events can increase soil erosion and contribute to hydraulic and airborne transport of turbidity-causing sediments into waterways. In steeper, forested headwaters in much of Oregon, canopies and associated detrital material make such areas resistant to erosion (Sorensen, et al. 1997).

In lakes and reservoirs, while coarser sediment inputs from streams generally settles out of the water column rapidly, small particles, such as clay, can remain in the water column for days and weeks (Kirk 1985). Moreover, sediment can be re-suspended by wind and waves (Arruda 1983). In addition, plankton blooms can result in turbid water in lakes (Utne-Palm 2002).

In estuaries, plankton blooms, bioturbation, soil erosion, and resuspension of sediments by wind, waves and currents can result in turbidity (Wilber and Clarke 2001). Wilber and Clarke (2001) summarize studies of natural suspended sediment levels reaching as high as 10,000 mg/L due to climactic events. Turbidity in estuaries can be influenced by wind speed, particle size and wave action (Cyrus & Blaber 1987). Tidal influences can cause a great fluctuation in suspended sediment-caused turbidity (Wilber and Clarke 2001).

Anthropogenic Sources of Turbidity

The following section presents literature regarding practices that can contribute to anthropogenically increased turbidity. With proper best management practices and/or treatment, such sources can be controlled, preventing significant impacts to beneficial uses. However, if done improperly or without the right safeguards in place, these sources can be significant contributors to increased turbidity levels.

Agriculture

Agricultural practices without proper BMPs have been shown to increase suspended sediment loads into rivers, reservoirs, and lakes that can contribute to increased turbidity levels (Arruda 1983). In Oregon, studies have shown that erosion from cropland is a major contributor to suspended sediment. In 2003, water (sheet and rill) erosion on cropland was estimated to be 5.8 million tons in Oregon (NRCS 2007). A DEQ study found that more than half of rivers near agricultural land in Oregon showed suspended sediment (and, presumably, turbidity) in mid- to most-disturbed conditions (Mulvey, et al. 2009). This is consistent with a 1997 study that found that agriculture in the Willamette Basin contributes the greatest amount of suspended sediment to the River (Miller, et al. 1997).

Construction and Urbanization

Urban development and construction of roads and buildings without proper BMPs have the potential to contribute substantial amounts of turbidity-causing sediments to Oregon's waters. Urban sites in the Willamette River basin contribute the greatest amount of suspended sediment to the river per acre (USGAO 1998). Construction sites without proper controls can contribute 35 to 45 tons of sediment to waters per year (USGAO 1998). Approximately 29% of streams near high and medium-intensity urban land uses and 17% of streams in low-intensity urban land uses were found to be in poor condition for total suspended solids in the Willamette Valley (Mulvey, et al. 2009). Total suspended solids at construction sites were significantly elevated compared to reference sites in a Colorado study (Cline, et al. 1982). However, one study found that road construction in forested watersheds of western Oregon was not a major influence on turbidity for domestic water sources (Grizzel and Beschta 1993). A study of the impacts of highways on sediment loads in the Navarro River watershed in California found that the major highway in the area contributed less than 1% of total sediment loads (Johnson, et al. 2002).

Direct Discharges (municipal, industrial and stormwater)

As compared to nonpoint sources, relatively little work has been done to examine the extent to which direct discharges impact turbidity levels. A longitudinal study on the Willamette River indicated turbidity peaks (an increase of 1-2 NTU compared to the nearest upstream station) downstream of a sewage treatment plant and a pulp mill, as well as from landfill and a pulp mill lagoon (Hughes and Gammon 1987). The National Council for Air and Stream Improvement (NCASI) has been measuring effluent turbidities from mills in Halsey and Springfield as part of a long-term receiving water study. Measurements from these mills taken from 1998 to 2001 indicate turbidities ranging from 7.7 to 33.1 NTU (NCASI 2002; 2003a; 2003b). Barter and Deas (2003) present the mean turbidity of different effluents measured with five different turbidimeters in New Zealand (Table A1).

Source	Mean NTU (±SD)
Stormwater	22.08±1.83
Fish processing effluent	110.70±11.01
Domestic waste water A	88.40±10.87
Domestic waste water B	170.51±18.89
Domestic waste water C	204.46±16.52
Dairy wash water	251.43±53.04
Apple processing effluent	305.90±65.09
Meatworks effluent	506.04±80.36

Table A1. Mean Turbidities from New Zealand effluents (from Barter and Deas 2003)

Dredging

Wilber and Clarke (2001) summarize potential impacts of dredging on suspended sediment concentrations. In general, concentrations of resuspended sediments vary depending on dredge and sediment types and environmental conditions at the time of dredging. Mechanical dredging (bucket and clamshell) increase sediment concentrations more than hydraulic dredging unless hydraulically pumped sediments are allowed to overflow. For a clamshell dredge, the maximum concentration of a sediment plume was 1,100 mg/L and extended as far as 1000 meters along the bottom. For hydraulic dredges, maximum concentrations are generally less than 500 mg/L and the plume usually extended 500 meters from the dredge (LaSalle 1990).

Forestry

Road construction and maintenance, harvesting, slash disposal, and site preparation associated with forest operations have the potential increase the availability of turbidity-causing sediment to streams (Everest, et al. 1987). A number of studies have looked at the potential for forest operations to increase turbidity. An eight-year study of the effects of logging in the Alsea Watershed found that suspended sediment levels were more than 200% higher in a basin that was clearcut without buffer strips than in an unharvested subbasin, and more than 50% higher in a basin that was clearcut with buffer strips than in the unharvested basin (Moring 1975). Studies of two subbasins (Kilchis and Tillamook) of the Tillamook Bay Watershed found a correlation between timber harvest and increased July-August turbidity (from 0.2 to 1.2 NTUs) (Naymik, et al. 2005; Ford and Rose 2000). ODF found in a two-year study of turbidity associated with wet weather use of roads on private and state forests in western Oregon that median turbidity increased below stream-crossing culverts (Mills, et al. 2003). Grizzel and Beschta (1983), in a study of 13 western Oregon municipal water sources, found that timber harvesting and road construction operations were not causing sustained increases in turbidity levels. Although forest operations preceding a storm appear to have triggered several landslides that impacted a municipal water source, turbidity increases were short-lived. A study in the Centennial Creek, British Columbia found that the main source of sediment was the main haul road and, to a lesser extent skid trails, landings, and clearcut stream channels (Slaney, et al. 1977).

Since the promulgation of Oregon's Forest Practices Act in the mid-1970s, best management practices on forested watersheds have evolved in response to various monitoring and research efforts. Many of these changes in forest practices have been directed at minimizing road-generated sediment, as well as sediment from other sources. Recently, Reiter, et al. (2009) found, in comparing different sections of the Deschutes River watershed in Washington State, that improved forest management, particularly with respect to road construction and maintenance, were correlated with declining turbidity levels.

Mining

There is a significant body of literature on the effects of placer and suction dredge mining on increased levels of suspended sediments and turbidity. Studies of placer mining indicate that its impact on turbidity can be long-lasting and acute. An early study of the effects of placer mining in the Rogue River Basin indicated that increased suspended sediment concentrations were only found on two of 13 mining impacted sites as compared to an un-mined site (Ward 1938). However, a study of placer mining on the Kenai Peninsula in Alaska found that high values of turbidity in one drainage were associated with mining activity (Huber and Blanchet 1992). Wastewater discharges from mining operations in that study averaged 167 NTU and reached as high as 1150 NTU. Turbidity that was measured daily in a placer mined-creek in Alaska averaged 727 NTU over three months; turbidity in an un-mined site averaged 1.3 NTU (Reynolds, et al. 1989). In a study of the same drainage, the average turbidity at five un-mined sites was approximately 7 NTU over a period of three weeks; average turbidity in eleven mined sites was 175 NTU over the same period (Scannell 1988).

As compared to placer mining, turbidity due to suction dredge mining appears to be short-lived and limited to a small stretch of stream where the dredge is operated. However, cumulative effects of several miners working in the stream, or of mining in the same area over weeks or months, are not clear. According to a literature review by Harvey and Lisle (1998), suction dredging can carry turbidity-causing fine sediment (clay, silt, and fine sand) downstream. Turbidity measurements taken above and below suction dredging on two California streams indicated a localized effect of suction dredge mining (Harvey 1986). At a study of suction dredge impacts in two Alaskan rivers, increased turbidity was noted downstream, but returned to upstream levels 160 meters downstream when an 8-inch dredge was used (Prussian, et al. 1999). A study of three suction dredge sites in Idaho showed that initial increases of turbidity were from 5-37 NTU above background, depending on the type of dredge used. Visible plumes were noted from 70-150 meters downstream, although the plumes were described as either "pulse-like" or "sporadic" (Stewart and Sharp 2003). A study of a 2.5-inch suction dredge in Montana showed increased suspended sediment levels immediately downstream of the dredge; these levels returned to normal within 11 meters downstream (Thomas 1985).