Washington State
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Establishing Appropriate Benchmarks for Site Development by Documenting Successional Characteristics

Phase 2:

Benchmarks for Stand Development of Forested and Scrub-Shrub Plant Communities at Wetland Mitigation Sites in the Lowlands of Western Washington

Mark T. Celedonia

Washington State Department of Transportation
Roadside and Site Development Unit
Olympia, Washington

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by:

Mark T. Celedonia
Washington State Department of Transportation
Environmental and Engineering Service Center
310 Maple Park Ave., SE
Olympia, WA 98504

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EXECUTIVE SUMMARY

Establishing reasonable and achievable success standards for wetland mitigation projects is currently hindered by a lack of data regarding the development of desired features at mitigation sites. In order to help bridge this gap, this study documented and evaluated features of 29 forested and scrub-shrub plant communities established at mitigation wetlands in western Washington. The main purpose of this study was two-fold: 1) document structural characteristics of woody plant stands at completed mitigation projects; and, 2) use this information to recommend reasonable and achievable benchmark standards that may be used for evaluating success of future mitigation projects.

Main objectives of the study were:

- Document aerial cover of native woody species and identify a benchmark standard for time to achieve 80% aerial cover;

- Document and evaluate abundance of woody nonnative invasive species and reed canarygrass, and propose benchmark standards as appropriate; and,

- Document and evaluate other structural attributes, and propose benchmark standards as appropriate. These included stem density of woody species, various measures of species richness and dominance, and establishment of planted and volunteer species.

These specific attributes were chosen because they are often used to help evaluate mitigation success in western Washington and/or because they represent basic stand development characteristics. This study did not attempt to identify an entire set of attributes that should be used to evaluate mitigation success, nor did it seek to recommend, identify or evaluate the appropriateness of specific attributes for evaluating wetland functions or mitigation success.

Mitigation sites between 6 and 11 years of age were evaluated for these purposes. *Time-series curves* were constructed from the data to evaluate age-related change in certain attributes. These curves provided pertinent information in evaluating stand development and proposing benchmark standards. Other relationships were evaluated independent of site age. These included: 1) influence of canopy cover and woody plant stem density on reed canarygrass cover; and, 2) influence of planting density on various plant community features. Results were used to identify benchmark standards and to consider management implications.

For aerial cover of native woody species, year 8 is the proposed benchmark standard for attaining 80% aerial cover. The time-series curve showed an increase in aerial cover until year 8, at which point 80% median aerial cover was reliably achieved.

Two related factors were identified that may help contribute to rapid establishment of abundant canopy cover. These included: 1) current densities ≥2,100 st/ac (4.6 ft oc) for trees and shrubs ≥2 m tall; and, 2) woody species planting densities ≥3,000 st/ac (3.8 ft oc), although this could not be confirmed. The former may represent a minimum post-installation survival density, while the latter requires further study. It is important to consider, however, that high stem densities could delay the development of other potentially desirable features, including plant maturity, emergence of a forested canopy, and vertical stratification. These relationships should be further studied.

For woody nonnative invasive species, the proposed benchmark standard is ≤5% aerial cover during years 6-11. These species were not subject to management controls after year 5. However, no age-related trend was detected during years 6-11, and levels did not exceed 5% median aerial cover during this time. Established stands of native woody species thus appeared capable of maintaining low levels of woody nonnative invasive species during years 6-11 without management intervention

For reed canarygrass, a benchmark standard was not proposed. No age-related trend was detected during years 6-11, however levels fluctuated between 1-22% median aerial cover during this time. Reed canarygrass was not subject to management controls after year 5. It was not certain whether higher levels were present at the end of year 5, or developed afterward. Due to these factors, a reliable benchmark standard could not be identified.

Reed canarygrass cover was related to stem density of trees and shrubs <6 m tall, but not canopy cover. This has two important implications. First, facilitating a dense initial layer of young woody plants may help prevent severe infestations early in the life of the site. However, this may also adversely impact development of other desirable attributes. The second implication is that stands in later stages of development may be subject to re-infestation. This is especially true when sparse understory is left beneath a vertically expanding canopy, which is common in stand development.

There were indications that planting density may have influenced extent of reed canarygrass proliferation, although this relationship could not be confirmed. Preliminary indications suggest that planting densities ≥3,000 st/ac (3.8 ft oc) may be optimum, however this requires further study.

For stem density of woody species, a benchmark standard was not proposed. Density is generally not used to evaluate mitigation success or wetland function. In addition, not enough information is known about how different stem densities affect the interaction between desirable stand attributes. This is an area that should be further explored.

Density of trees and shrubs increased from planting to present, but any age-related change from years 6-11 could not be detected. Failure to detect general age-related change was likely due to inadequacy of the methods used. The following relationships were observed for common planting densities (≤3,200 st/ac; 3.7 ft oc): 1) stem density generally increased by factors of 1.1-3.1 from planting to present; and, 2) current density was partially dependent upon planting density.

For woody species richness and dominance during years 6-11, the following benchmark standard were proposed:

- richness of woody species = number of planted species
- richness of tree and shrub species = 4 tree and 6 shrub species
- richness of dominant species:
  - = 4 tree and 3 shrub species at ≥1% aerial cover/species
  - = 2 tree and 2 shrub species at ≥5% aerial cover/species
  - = 2 tree and 1 shrub species at ≥10% aerial cover/species

The proposed benchmark standards are believed to be both reasonable and achievable using common mitigation practices. However, they may become outdated as mitigation science and practice evolve. Periodic review and revision are thus important to ensure standards are appropriate for current practice.

Concepts of forest stand development provided a useful framework for understanding results and proposing benchmark standards. Specifically, these concepts provided a fuller understanding of the results and allow consideration of the broader implications of proposing benchmark standards for specific attributes. Application of such concepts to wetland mitigation should be further considered and explored.

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1.0 INTRODUCTION

1.1 Background

In 1997, a group of wetlands professionals from various state, federal, and private entities within the Pacific Northwest began meeting to help clarify and refine the use of success standards in wetland mitigation (Ossinger 1999). This group was called the *Success Standards Work Group* (SSWG). Among other objectives, the SSWG sought to identify and promote reasonable and achievable success standards, and to suggest research priorities. Vegetation was recognized as a top research priority. This was primarily because: 1) specific features of vegetation are some of the most commonly used indicators of success; and, 2) little has been documented on the characteristics and development of vegetation at natural or mitigated wetlands in the Pacific Northwest.

In order to contribute to this process, the Washington State Department of Transportation (WSDOT), with help from two U.S. Environmental Protection Agency (EPA) Wetland Program Development Grants, initiated a two-phased study of vegetation at wetland mitigation sites. This study was undertaken with the intent of surveying current knowledge of wetland vegetation development, and producing reasonable and achievable benchmark standards for measuring the success of vegetation at mitigation projects. Phase I of the study was completed in October 1999 (Lindstrum and Maurer 1999), and accomplished the following objectives: 1) reviewed and summarized current literature and research on wetland plant succession and mitigation performance standards; 2) established site selection criteria, and identified potential sites for further study; 3) proposed a methodology for studying these sites; and, 4) performed a pilot study of potential monitoring methods on a sample of the identified sites.

Phase II began in February 2001, and was to include: 1) collection of background information on potential study sites, including mitigation plans, planting plans, as-built reports, monitoring reports, and documentation of corrective action; 2) final site selection; 3) where necessary, gaining right of entry permission from landowners; and, 4) collecting and analyzing data, and interpreting and reporting the results. Due to various circumstances, the scope of Phase II changed substantially from that described in the Phase I report. The reader is therefore advised to rely on the information contained herein for scope and methodologies relevant to Phase II.

1.2 Phase II Study

Phase II of the study focused on forested and scrub-shrub mitigation wetlands in western Washington. **The main purpose of this study was two-fold: 1) document structural characteristics of woody plant stands at completed mitigation projects; and, 2) use this information to recommend reasonable and achievable benchmark standards that may be used for evaluating success of future mitigation projects.**

Priority was placed on documenting and evaluating structural attributes that: 1) are commonly used as indicators of mitigation success in Washington State; and, 2) provide a more complete understanding of site development.

Main objectives of the study were to:

- **Document aerial cover of native woody species[1] and identify a benchmark standard for time to achieve 80% aerial cover;**

- **Document and evaluate abundance of woody nonnative invasive species[2] and reed canarygrass, and propose benchmark standards as appropriate; and,**

- **Document and evaluate other structural attributes, and propose benchmark standards as appropriate. These included stem density of woody species, various measures of species richness and dominance, and establishment of planted and volunteer species.**

These are discussed in more detail below. It should be noted that these attributes are often used in conjunction with others in evaluating wetland functions and mitigation success. **<u>This study did not attempt to identify an entire set of attributes that should be used to evaluate mitigation success, nor did it seek to recommend, identify or evaluate the appropriateness of specific attributes for evaluating wetland functions or mitigation success.</u>**

One of the most common features used to judge success of forested and scrub-shrub wetland zones is aerial cover of native woody plant species. Often times, 80% aerial cover is required by a site's fifth year[3] for it to be deemed successful. However, indications from mitigation professionals suggest that most stands do not achieve 80% cover by year 5, but likely do soon thereafter. Thus, this study sought to identify a benchmark standard for time to achieve 80% aerial cover of native woody species under common management regimes[4] and without consideration for other stand attributes (e.g., impact of rapid canopy cover establishment on other stand attributes such as plant maturity, emergence of a forested canopy, and vertical stratification).

Standards for undesirable species are also common. This study chose to evaluate abundance of: 1) woody nonnative invasive species; and, 2) reed canarygrass, perhaps the most problematic undesirable herbaceous species. This study was designed to shed light

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[1] Native woody species encountered in this study are identified in Appendix C.
[2] Woody nonnative invasive species encountered in this study are identified in Appendix C.
[3] The age of a site is defined as the number of growing seasons the site has experienced after planting, including the one during which the current data were collected. For example, a site planted during the winter or spring of 1994, and where data were collected during summer 2001, was considered a year 8 site, or in its 8th year.
[4] Since this study used completed projects as the basis for establishing benchmarks, results may be perceived as representing old or outdated management practices and not common *current* practices. This is addressed in Section 2.1.2.

on what occurs after the typical 5-year monitoring period when human efforts to control these species are no longer present. Thus, the main questions this study sought to address with respect to reed canarygrass and nonnative woody species were: 1) what levels of abundance generally exist or develop after 5 years when control efforts are absent; and, 2) do these levels change with time? Benchmark standards were proposed as appropriate for monitoring periods that extend beyond 5 years.

Other stand attributes were also evaluated for their usefulness as benchmark standards, as well as to lend insight into stand development progress. These included various measures of species richness and dominance, as well as stem density of trees and shrubs. Presence and abundance of planted and volunteer species were also evaluated.

Mitigation sites ranging in age between 6 and 11 years were evaluated for these purposes. Vegetation data were collected from 29 stands of woody vegetation within 24 palustrine wetland mitigation sites located in the lowlands of western Washington (Hruby et al. 1999). Only *successfully established* [5]stands were surveyed. Data collection and analysis included:

- <u>Quantitative data on stand attributes</u> – Quantitative data on the following attributes were collected from each stand.

    - aerial cover of native woody species;
    - stem density of woody species;
    - abundance of woody nonnative invasive species;
    - abundance of reed canarygrass;
    - richness of woody species;
    - richness of tree and shrub species; and,
    - richness of dominant woody species.

- <u>Influence of site age on stand attributes</u> – The influence of site age on stand attributes was evaluated. All attributes for which data were collected were evaluated in this manner, except for richness of tree and shrub species, and richness of dominant species. *Time-series curves* similar to the performance curves described by Kentula et al. (1992) were constructed for these purposes.

- <u>Other stand features and relationships</u> – Other stand features and relationships were evaluated independent of site age. These included presence and abundance of planted and volunteer species, as well as various relationships between canopy cover, stem density, reed canarygrass cover, and planting density. For example, one relationship that was assessed was that between canopy cover and reed canarygrass: do stands with more woody plant cover have less reed canarygrass, regardless of site age? These are discussed in more detail in Section 2.2.5.2.

[5] *Successfully established*, as used in this study, is defined in Section 2.2.2, and does not imply that success standards were met.

1.3 Ecological Framework

Basic principles of forest ecology provided a conceptual framework used in designing the study and interpreting the results. Specifically, pertinent concepts were derived from the study of *stand development* - how a group of trees and associated vegetation changes over time. These concepts and their application to wetland mitigation are discussed briefly below.

1.3.1 Stand Development Concepts

The following concepts of stand development were derived primarily from Oliver and Larson (1996), and supplemented by Barnes et al. (1998). Xue and Hagihara (1999) provided confirmation that the decrease in density observed during the stem exclusion stage is driven by competition for light.

> ***Stand* –** "A spatially continuous group of trees [or shrubs] and associated vegetation having similar structures and growing under similar soil and climatic conditions" (Oliver and Larson 1996).
>
> ***Stand structure* –** "The physical and temporal distribution of trees [or shrubs] and other plants in a stand" (Oliver and Larson 1996). This can include one or more of the following: species, vertical and horizontal spatial patterns, plant size, plant ages.
>
> ***Stand development* –** Changes in stand structure over time, including those changes that occur following a disturbance. Forest ecologists generally recognize four stages of stand development: stand initiation, stem exclusion, understory reinitiation, and old growth:
>
>> ***Stand initiation* –** Following a major disturbance (e.g., fire), new individuals and species colonize the disturbed area. During this period, plant density is generally high and continues to increase until the available growing space is occupied. Most often, the limiting resource is light, although other resources can become limiting. Stand initiation thus continues until the canopy closes and new individuals can no longer establish. The stand initiation stage is often characterized by high numbers of plant species.
>>
>> ***Stem exclusion* –** Once the canopy closes, new individuals can no longer establish and existing individuals compete for light in order to continue growing. As a result, density-dependent mortality begins to occur. Individuals with a competitive advantage take over the growing space of others, causing the outcompeted individuals to die. This stage is thus marked by a decrease in stem density. One result of this thinning process is a sparsely vegetated understory beneath a thick and vertically expanding canopy. The stem exclusion stage is often characterized by a decrease in plant species richness.

***Understory reinitiation* –** As the overstory ages, the canopy begins to thin somewhat allowing more light into the understory. Shade-tolerant trees, shrubs and herbs then colonize the forest floor. Species richness increases over that generally observed during stem exclusion.

***Old-growth* –** As individuals in the overstory mature, reproduce, age and die, new individuals from the understory grow to replace them. The results is an interspersion of canopy cover and canopy gaps, young and old individuals, and other properties.

***Stagnation* –** Condition whereby tree and shrub growth is repressed or halted due to equivalent competition for resources among crowded individuals. Stagnation is never absolute; that is, trees and shrubs do not stop growing, they just grow more slowly. Stagnation commonly occurs in denser stands. One result of stagnation is a slowing in the overall maturation of the stand. Thus, a stand affected by stagnation may be characterized by smaller individuals, a lower canopy, and less vertical stratification than a similarly aged stand unaffected by stagnation.

1.3.2 Application of Stand Development Concepts to Wetland Mitigation

Stand development concepts may be applied to any stand of woody vegetation. Applied in this study, they provide a fuller understanding of results and allow consideration of the broader implications of proposing benchmark standards for specific attributes. Although derived primarily from forest stand development, the same concepts are applicable to shrub communities (Oliver and Larson 1996).

It should be kept in mind that stand development concepts are intended more as a conceptual framework than hard and fast rules. Stands may resemble quite closely the idealized development patterns discussed above, or they may fit only loosely. Stands comprised of heterogeneous mixtures of species within interspersed growing conditions - which is often the case with wetland mitigation sites - are more likely to show variability in development. A more careful application of the concepts in these situations is more helpful than not applying them at all.

It could be argued that most if not all stand attributes associated with desired wetland functions are provided by more mature stands in later stages of development. To the extent that this is true, a primary objective of mitigation efforts is to expedite the development of a young, newly installed stand through the early stages as rapidly as possible. These early stages are: 1) characterized by dramatic and fluctuating structural attributes; and, 2) may last anywhere from several years to several decades or more. In addition, attributes displayed while the stand is in one stage may impact stand development in later stages. For example, higher stem densities during stand initiation are more conducive to stagnation, and thus a prolonged stem exclusion stage and delayed understory reinitiation. Benchmark standards established with consideration for stand development patterns are thus more informed and meaningful.

2.0 METHODS

2.1 General Methodology

2.1.1 Selection of Stand Attributes to Measure and Evaluate

Success standards often use morphological characteristics (e.g., aerial cover of trees and shrubs) as a measure of wetland function (e.g., habitat suitability). A variety of vegetative attributes have been proposed and used for evaluating wetland functions (Azous et al. 1998; Hruby et al. 1999; Cooke 2000; Null et al. 2000). It was beyond the scope of this study to assess and provide benchmarks for all attributes related to these functions. The following criteria were used in selecting the stand attributes to evaluate: 1) commonly used indicators of success; and, 2) other attributes that relate to stand development progress. This resulted in selection of the following stand attributes to measure and evaluate:

- aerial cover of native woody species;
- stem density of woody species;
- abundance of woody nonnative invasive species;
- abundance of reed canarygrass;
- richness of woody species;
- richness of tree and shrub species; and,
- richness of dominant woody species.

2.1.2 Identifying Benchmark Standards

This study sought to recommend benchmark standards that are both *reasonable* and *achievable*. In order to best accomplish these goals, methods were selected to produce benchmarks that: 1) are based on actual results of mitigation practice; and, 2) are better than the absolute minimum that can be achieved, yet are not so high as to be unattainable. **Benchmark standards were thus based on *completed projects* that met no less than a *minimal level of success*:**

- Using *completed projects* as a basis for establishing benchmarks helped ensure that standards would be *achievable* by avoiding the problems associated with other methods. Specifically, more natural or undisturbed systems used as references often provide unachievable standards in evaluating short-term (5-10 years) success of restored, created or enhanced systems (Cairns 1990; Simenstad and Thom 1996; Lockwood 1997; Zedler and Callaway 1999; Kentula 2000).

- The *minimal level of success* required of completed projects was simply that a stand of healthy and growing native trees and shrubs was established[6]. It was assumed that: 1) such established stands represented the minimum and maximum of what can be accomplished with common mitigation practices; and, 2) benchmarks based on some middle ground between the minimum and maximum would be *reasonable*.

Medians were selected as the basis for recommending benchmark standards. That is, for a given attribute, the median of all stands surveyed was believed to best represent a *reasonable* and *achievable* benchmark standard. The median was selected for two main reasons. One is that it was believed to best represent the middle ground between the minimum and maximum discussed above. By definition, the median conveys the exact level where the top 50% of sites are separated from the bottom 50%. In addition, it may provide a better indicator of central tendency (i.e., the middle ground) than the mean when extreme measurements and skewed distributions are present.

The second reason for selecting the median as the basis for benchmarks was to accommodate improvements in mitigation practice. One possible disadvantage of using completed projects as a standard is that mitigation practices have been improving over time. Standards based on completed projects may, therefore, not be adequate to evaluate results of new and improved practices. This is an inherent consequence of using completed projects to set standards. Selecting the median as the basis for benchmarks compensates for this by "raising the bar" and suggesting that improvements in mitigation practice should allow *all* new projects to be at least as successful as the *upper half* of minimally successful completed projects.

2.1.3 Management Implications

Management implications of the findings were explored and discussed. This was done to identify some of the possible factors that may have led to the characteristics exhibited by the more successful sites. This is critical considering that these more successful sites formed the basis for recommending benchmark standards.

A thorough examination of all possible factors contributing to the more successful mitigation outcomes was beyond the scope of this study. **Therefore, the management implications discussed are offered more to generate thought, discussion and research than to recommend precise and proven practices for achieving specific results.** Furthermore, the management implications offered were based only on achieving the specific attribute(s) discussed, and generally did not include any consideration of monetary costs or effects on other wetland functions and features.

[6] Specific criteria for determining whether stand met this minimal level of success are detailed in Section 2.2.2.

2.2 Research Methods

2.2.1 Historical Information Retrieval

Historical information was collected on the potential study sites identified during Phase I. Such information included mitigation plans, planting plans, as-built reports, monitoring reports, and documentation of corrective actions. The following sources were reviewed for pertinent documents: U.S. Army Corps of Engineers (USACE) Section 404 permit files; Washington State Department of Ecology (WSDOE) Section 401 database; WSDOT wetland mitigation project files; relevant county and city government files; and files of the various consultants involved in the planning, constructing and monitoring of the study sites.

2.2.2 Final Site Selection

Phase I identified 35 wetland mitigation projects in western Washington as possible study sites (Lindstrum and Maurer 1999). Of these, sites meeting the following criteria were selected for study:

- <u>Wetland mitigation intent</u> - The site was intended to serve solely as wetland compensatory mitigation. Sites intended to serve other or dual purposes were excluded (i.e., bioswales, stormwater detention ponds).

- <u>Forested and/or scrub-shrub zones</u> - Project documentation indicated at least a portion of the site was intended to develop into a forested or scrub-shrub wetland, as defined by Cowardin et al. (1979). Some projects called for “riparian” or “forested/scrub-shrub” wetlands, which are not defined in Cowardin et al. (1979). These were included in the study.

- <u>Successful establishment of native woody vegetation</u> – A site visit verified that native woody vegetation: 1) occurred within the approximate area of the intended zone; and, 2) appeared to be healthy and growing and/or reproducing. Sites were excluded if they contained minimal cover of woody vegetation, and where woody plants were generally absent, stunted and sparse, or severely stressed. Although somewhat subjective, there was very little uncertainty in distinguishing successfully established stands versus failed or struggling stands of woody vegetation in the field.

- <u>Wetland established</u> – Site visit verified that the area to be studied exhibited appropriate wetland features. This was accomplished by implementing basic wetland delineation techniques as described in the USACE Wetlands Delineation Manual (Environmental Laboratory 1987). Areas that did not exhibit sufficient wetland character were excluded.

- Site access – Where necessary, written Right of Entry permission from the landowner was secured prior to visiting the site.

Twenty-four of the 35 sites identified during Phase I failed to meet one or more of these criteria (Table 1). Due to this high rejection rate, additional WSDOT sites not considered during Phase I were evaluated for inclusion in the study. Time constraints prohibited other public or private mitigation projects from being considered. Thirteen WSDOT sites were added, bringing the total number of study sites to 24. A detailed list of selected study sites and associated information, including location, age, mitigation type, Ecology rating (WSDOE), and hydrogeomorphic class, is provided in Appendix A.

2.2.3 Defining Research Units

The specific area of a site where data were to be collected and evaluated was termed a *research unit* (RU). An RU generally consisted of either: 1) the entire wetland portion of the site, if it was intended to be forested or scrub-shrub; or, 2) a forested or scrub-shrub zone of a larger wetland containing multiple Cowardin et al. (1979) classes. **Only wetland areas were included in RU’s - buffers and other upland areas were not included.** In most cases, each site was represented by one RU. Four sites provided multiple RU’s: three sites provided two RU’s each, and one site provided three RU’s.

A study site provided more than one RU under two possible conditions. The first occurred when two or more distinct plant communities existed adjacent to one another, with each intended as a different zone. For example, a clearly defined strip of willows (*Salix* spp.) designated as a riparian zone might constitute one RU, while an adjacent forested zone of red alder (*Alnus rubra*) might comprise another. The second condition occurred when one stand of woody plants was separated from another by a substantial physical feature, such as a large emergent or open water zone, or an upland area. Several

Table 1. Rejection of potential study sites identified in Phase I, including number of sites and reasons for their rejection.

Number of sites rejected	Reason for rejecting
11	site documentation did not indicate a clearly intended forested or scrub-shrub wetland zone
6	written right of entry was denied or not obtained in time
2	intended for purposes other than or in addition to wetland mitigation
2	woody vegetation failed to establish
2	historical project documentation could not be located or obtained
1	consisted of hydrologic manipulation to existing forested wetland

sites were designed to have separate stands of woody plants in different areas of the site. However, many of these sites provided no more than one RU each due to failure of other stands to successfully establish and/or to exhibit sufficient wetland character.

A total of 29 RU's were evaluated for this study. A summary of RU distribution by age, location, size, Cowardin et al. (1979) class, and other features is provided in Table 2. A detailed list of features for each RU is provided in Appendix A.

Table 2. Distribution of research units by age, size, location, mitigation type, Cowardin et al. (1979) class, WSDOE rating, HGM class, and surrounding land use. Numbers in parenthesis indicate the number of study sites represented, if different than the number of RU's.

Criteria	No.	Criteria	No.	Criteria	No.
Age (year)		**Location (county)**		**HGM Class[e]**	
6	7 (5)	King	15 (11)	Depressional	
7	5	Kitsap	4	closed	10 (9)
8	7 (6)	Snohomish	4 (3)	outflow	11 (10)
9	0	Clark	3	Riverine	
10	8 (6)	Lewis	1	flow-through	4 (2)
11	2	Pierce	1	impounding	1
Size (acres)		Whatcom	1	Slope	3 (2)
0.00-0.09	1	**Mitigation type[a]**		**Surrounding Land Use[f]**	
0.10-0.24	9	Creation	13 (11)	Developed	11 (10)
0.25-0.49	9	Enhancement	9 (6)	Rural	11 (8)
0.50-0.99	4	Restoration	7	Greenbelt	6 (5)
1.00-1.99	5	**Cowardin class[b,c,d]**		Natural-Forested	1
2.00-2.99	1	Forested	17	**WSDOE Rating[g]**	
		Scrub-shrub	12	Class 2	12 (10)
				Class 3	17 (14)

a Adapted from Gwin et al. 1999. Gwin et al. (1999) also include definitions for exchange and expansion. These were considered enhancement and creation, respectively.
b Cowardin et al. 1979.
c As specifically stated in the mitigation plan. It should be noted, however, that it was not uncommon to encounter plans indicating a scrub-shrub zone, yet planted copiously with species that often grow taller than the 6 m scrub-shrub threshold, such as *Salix lucida* var. *lasiandra* and *Salix sitchensis.*
d Some projects indicated "riparian" or "forested/scrub-shrub" wetlands. When this occurred, a Cowardin et al. (1979) class of forested or scrub-shrub was assigned based on which best represented the apparent intent of the mitigation plan.
e Hruby et al. 1999.
f See Section 2.2.4.1 for land use definitions.
g WSDOE 1993.

2.2.4 Observations and Data Collection

Quantitative vegetation data and other field observations were collected and recorded at each RU. A description of the types of data collected and the methods employed is provided below. Data were collected by one biologist with assistance from two college interns between June 25 and September 6, 2001.

2.2.4.1 Ratings, Classifications, and Other Qualitative Observations

A variety of qualitative data were collected and recorded at each RU. These included observations on soils, herbaceous vegetation, hydrology indicators, hydrogeomorphology, Washington State wetlands class, and surrounding land use:

- Hydrology indicators - Each RU was surveyed for the presence of hydrology indicators, as described in the USACE Wetlands Delineation Manual (Environmental Laboratory 1987).

- Herbaceous vegetation - Herbaceous plant species within each RU were identified and recorded. Approximate aerial cover for each species was occularly estimated.

- Soils - Two or more soils pits were excavated within each RU. Pits were excavated to a depth of 45 centimeters or more. Soil horizons were identified, measured and recorded. Each horizon was assessed for: approximate content of clay, silt, sand, organic matter, and moisture; presence of redoximorphic features (mottling); and Munsell® soil color (GretagMacbeth Corporation 2000) of matrix and any redoximorphic features.

- Hydrogeomorphic (HGM) classification - Each RU was classified according to hydrogeomorphic properties (Hruby et al. 1999). Classifications were accomplished using historical site documentation, including grading plans and topographic maps, and visual observations during site visits.

- Washington State wetlands class - Each site was rated according to WSDOE's Wetlands Rating System (WSDOE 1993). Ratings were completed during site visits.

- Surrounding land use - Surrounding land use at each site was described using aerial photos[7] and visual observation. Predominant surrounding land use was characterized by general appearance as follows: *Developed* - urban, commercial,

[7] Aerial photos were obtained from the WSDOT Aerial Photography Lab. Photos were taken during the summer of 2000 as part of routine WSDOT operations, not specifically for this study. Photos existed for 15 of the 24 study sites.

industrial, high density residential, or other heavily impacted areas; *Greenbelt* - natural areas, parks, or open spaces within developed areas; *Rural* - agricultural areas, low density residential, or other more lightly developed areas; *Natural* - relatively undisturbed lands.

2.2.4.2 Quantitative Data

Vegetation data were collected using methods outlined in Elzinga et al. (1998). Each RU was sampled in its entirety. Sampling was performed by first establishing a baseline across the entire RU length. Transects were then randomly located along this baseline. Transects were spaced no closer than 2 m apart, and no farther than 10 m apart. Transects spanned the entire width of each RU. Systematic random sampling designs were used in all cases but one, where peculiarities of the site warranted a restricted random sample.

Vegetation data were collected along each transect as follows:

- Woody vegetation aerial cover - Aerial cover data were collected along the entire length of each transect using the line intercept method.

- Woody vegetation stem density - Density data were collected within quadrats defined by transect length and 1 m width. Individual trunks or stems emerging from the soil served as the counting unit. Trunks, branches or stems connected to a common base at, above, or near the ground surface were considered one stem. Stems were categorized by height as follows: <1.0 m; 1.0-1.9 m; 2.0-5.9 m; and ≥6.0 m.

- Reed canarygrass (*Phalaris arundinacea*) aerial cover - Aerial cover data were collected along each transect using the point intercept method, with points systematically located along the transect. Points were spaced at 1.6 ft intervals.

Transects within a given macroplot were usually unequal in length. Therefore, randomly selected segments of transects were excluded in order to create *sample units* of equivalent length. The desired minimum sample unit length was 10 m, and the desired minimum number of sample units was twenty per macroplot. Sample units were deemed equivalent if their lengths were within ±10% of one another. Two or more sample units were occasionally created from one transect. In most cases, data were collected in the field along the entire length of each transect, with sample unit establishment and random exclusions performed later with computer assistance.

Sample units were averaged to provide representative values for each feature measured within the macroplot. These means were used to represent RU’s in analyses.

Total woody species richness for each RU was determined during site visits. Field personnel recorded all species observed while performing site walk-throughs and while sampling for other attributes.

2.2.5 Analysis

Two basic types of analyses were performed: 1) influence of site age on stand attributes (e.g., native woody plant cover, reed canarygrass cover); and, 2) evaluation of stand features and relationships independent of site age. These are described in more detail below.

2.2.5.1 Influence of Site Age on Stand Attributes

Time-series curves were constructed and used to evaluate influence of site age on the following attributes: native woody cover, nonnative woody cover, density of trees and shrubs <6 m tall, density of trees and shrubs ≥6 m tall, and reed canarygrass cover. Curves were constructed for each attribute by first grouping RU means by age. For example, mean values of native woody cover from all year 6 RU's were grouped, as were those from year 7, year 8, and so forth. Age classes represented by fewer than 5 RU's were lumped with an adjoining year[8]. For each age group, the median (M) and the interquartile range[9] (IQR) were calculated and plotted to provide resistant measures[10] of central tendency and variability. This method of constructing time-series curves is similar to one described by Kentula et al. (1992) for generating performance curves.

Time-series curves were examined for discernable patterns consistent with reasonable expectations. Hypothesized patterns were tested statistically using a Kruskal-Wallis test with either a directional or nondirectional alternative hypothesis as appropriate (Zar 1999; Sheskin 2000). Results of these and other statistical analyses were deemed significant at $p \leq 0.05$ and weakly significant at $p \leq 0.10$.

These methods risk falsely identifying or obscuring age-related trends. For example, Kentula et al. (1992) indicate that general mitigation design practices may change over time, resulting in a curve that might be more representative of design changes than of actual time-related change in the attribute measured. Thus, in order to obtain reliable results, an *assumption of accurate representation* must be satisfied. This assumes that each age group accurately represents the general time-related nature of the attribute being measured. Substantial violation of this assumption may be guarded against by including

[8] Year 11 had a sample size of only 2 RU's; therefore, these data were lumped with year 10 data. Year 9 was not represented in the data.
[9] The *interquartile range* (IQR) is the range of data that extends from the 25th percentile to the 75th percentile when the data are arranged sequentially. Thus, the lowest 25% of data lie below the IQR, the highest 25% lie above the IQR, and the middle 50% of data lie within the IQR.
[10] *Resistant* means less affected by extremely high or extremely low points within the data set (Zar 1999). Less resistant statistics (e.g., mean, standard error) used under the same circumstances may be misleading.

only similarly designed sites and by maintaining regional specificity, among others (Kentula et al. 1992).

2.2.5.2 Other Stand Features and Relationships

Other stand attributes, such as species richness, were presented to provide a general idea of vegetation structure and dynamics at the RU's surveyed. The median (M) and interquartile range (IQR) of all RU's were calculated to provide resistant measures of central tendency and variability. The IQR was used to represent the range exhibited by most RU's, unless otherwise noted.

Relationships between various vegetative features were also assessed. Most often, these consisted of using simple linear regression or correlation (Zar 1999), as appropriate, to assess the relationship between two attributes such as reed canarygrass cover and canopy cover. Only those analyses yielding significant results or otherwise providing insight into wetland vegetation development were reported.

For these purposes, the following additional parameters were tabulated from the data:

- planting density of woody species (d_P);
- number of woody species planted (N_p);
- number of woody species retained (i.e., planted and still present) (N_r);
- number of woody species failed (i.e., planted but no longer present) (N_f);
- number of volunteer woody species (N_v).

Historical site documentation was used to derive d_P and N_p. Estimates of d_P included supplemental plantings installed in years following the initial planting. Tabulation of N_p included all species planted within the RU, as well as species planted elsewhere on the mitigation site that appeared within the RU during the survey. Estimates of N_r were determined by subtracting N_f from N_p. Volunteer species were assumed to be all species observed within the RU that were not planted anywhere on the mitigation site.

Planting densities and species planted could not confidently be determined for 2 RU's due to inadequate historical documentation. Also, cover data for individual species could not be calculated for 3 RU's due to recording oversights. Where appropriate, these RU's were excluded from analyses.

3.0 RESULTS

Results are summarized and presented below. A detailed list of results for each RU is provided in Appendix B.

3.1 Aerial Cover of Native Woody Species

The time-series curve constructed for native woody cover (Figure 1) showed a trend consistent with the conclusion of stand initiation and transition into later stages of stand development. This curve indicated that aerial cover increases with age until year 8, then remains constant into years 10-11. Statistical analysis confirmed this pattern (Kruskal-Wallis, directional: $p = 0.0265$). This general pattern conformed with reasonable expectations for canopy cover expansion over a developing site: cover expands until available growing space is occupied and a relatively stable level of cover is achieved (Oliver and Larson 1996; Barnes et al. 1998).

Cover was related to stem density of trees and shrubs currently ≥2 m tall (Figure 2). A strong positive relationship was found for densities ≤2,100 stems/acre (st/ac) (linear regression: $p = 0.0004$, $r^2 = 0.5727$, n = 17). At densities >2,100 st/ac, 10 of 12 RU's had aerial cover ≥90%. These results suggest that a minimum density of 2,100 st/ac (4.6 feet on center [ft oc]) of taller growing shrubs and trees may be critical for achieving high levels of cover during years 6-11.

Cover may have also been related to planting density, although this relationship could not be confirmed. A direct comparison of cover versus planting density (Figure 3) showed

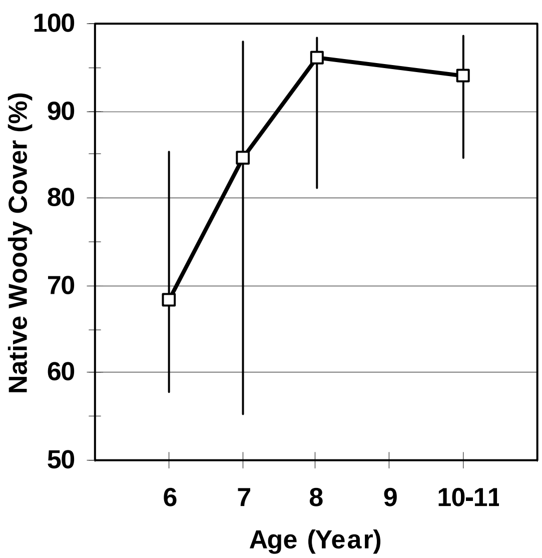

Figure 1. Time-series curve for aerial cover (%) of native woody species. Boxes (□) indicate the median of each age group; vertical lines represent the interquartile range.

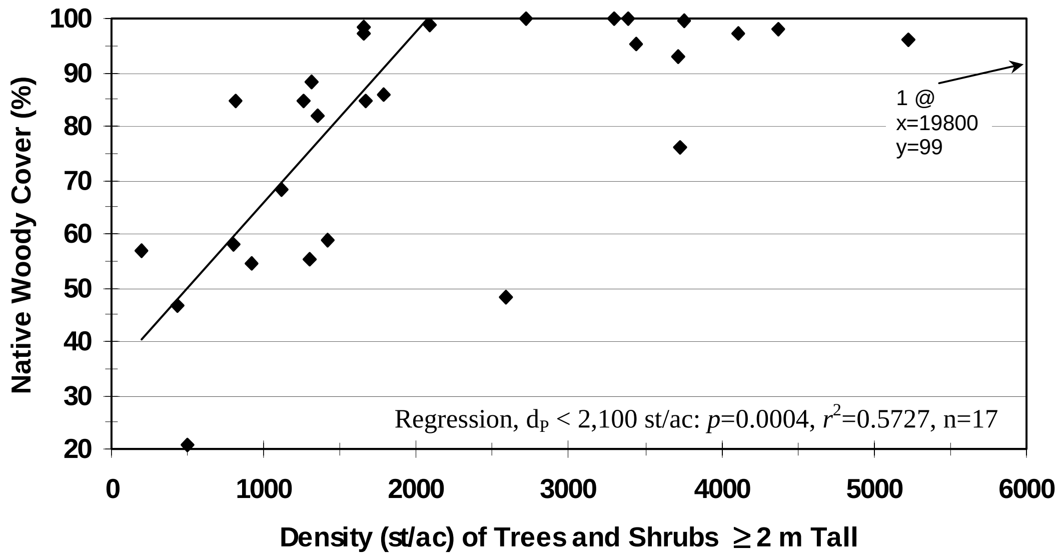

Figure 2. Aerial cover (%) of native woody vegetation versus stem density (st/ac) of trees and shrubs currently ≥2 m tall. Line indicates results of simple linear regression for stem densities ≤2,100 st/ac (4.6 ft oc).

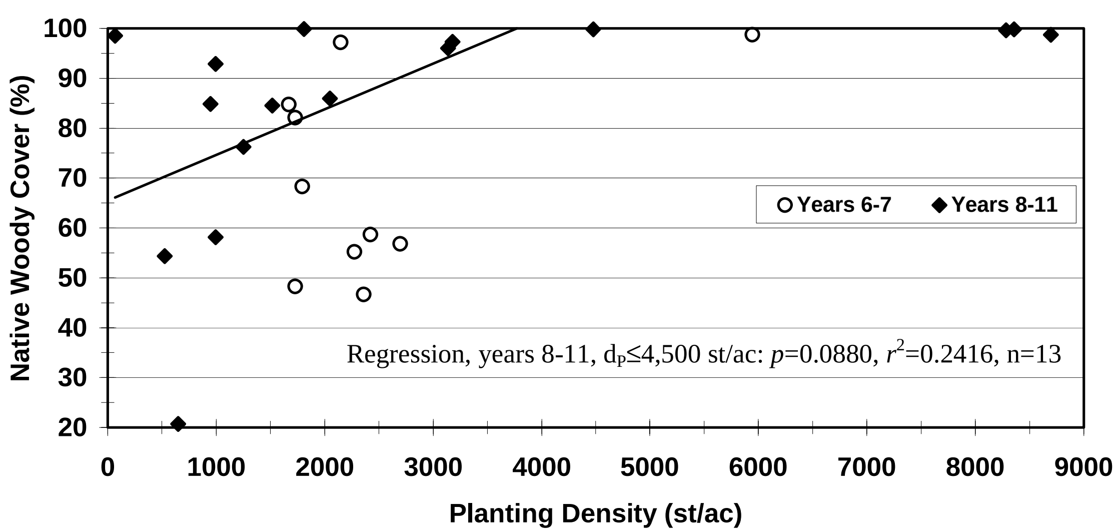

Figure 3. Aerial cover of native woody vegetation (%) versus planting density (st/ac) of trees and shrubs. The line indicates results of simple linear regression for RU’s in years 8-11 and with planting densities ≤4,500 st/ac (3.1 ft oc).

that high planting densities (≥3,200 st/ac; 3.7 ft oc) consistently yielded high levels of cover, although only 7 RU's were planted in this range. A weak positive relationship was observed for years 8-11 and planting densities from 100-4,500 st/ac (20.9-3.1 ft oc) (linear regression: $p = 0.0880$, $r^2 = 0.2416$, n = 13); however, these data appeared highly variable and no such relationship was observed for years 6-7.

Two other indicators suggested a possible connection between cover and planting density. These both involved indirect associations through tall (≥2 m in height) woody plant stem density, an apparent predictor of cover. First, density of 2 m and taller trees and shrubs showed a weak but direct dependence on planting density (linear regression: $p = 0.0528$, $r^2 = 0.1534$, n = 25) (Figure 4). These results were deemed inconclusive, though, due to: 1) the highly skewed nature of the data (n = 22 for $d_P < 4{,}500$ st/ac, and n = 3 for $4{,}500 < d_P < 8{,}700$ st/ac); and, 2) failure to detect significance after removing influential data points.

The second indicator was the significant relationship observed between planting density and current woody plant stem density (see below). Current stem density did not show any direct bearing on cover, but did show a strong correlation with density of trees and shrubs ≥2 m tall (linear correlation: $p < 0.0001$; $r = 0.6748$, n = 28). This implied a relationship from planting density through current stem density and tall plant density to cover. This was not deemed sufficient to confirm a relationship between cover and planting density, however.

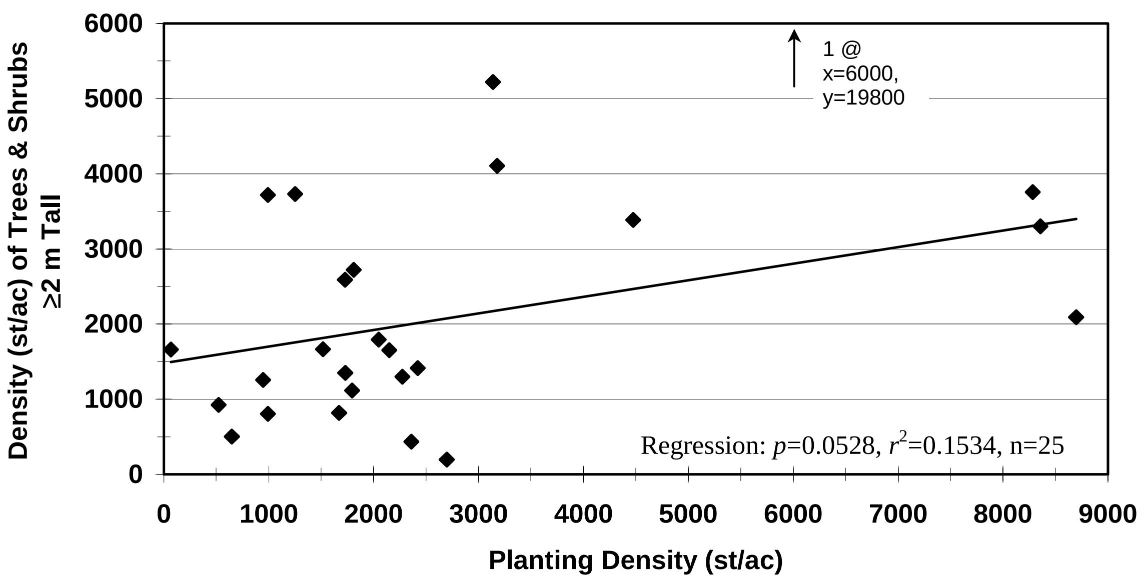

Figure 4. Current stem density (st/ac) of trees and shrubs ≥2 m tall versus planting density (st/ac). The line indicates results of simple linear regression, excluding the datum at (x=6000, y=19800).

3.2 Stem Density of Woody Species

3.2.1 Density of Trees and Shrubs <6 m Tall

Density of trees and shrubs <6 m tall appeared to be associated with planting density, but not with age. The time-series plot was suggestive of a possible age-related pattern, with annual medians increasing from year 6 through year 8, then decreasing in years 10-11 (Figure 5A). However, this trend closely followed that of median annual planting density, shown on the same plot. Further analysis found that, regardless of age, current density of trees and shrubs <6 m tall was significantly associated with planting density for RU's planted at ≤3,200 st/ac (3.7 ft oc) (linear regression: $p = 0.0131$, $r^2 = 0.2830$, n = 21) (Figure 6).

3.2.2 Density of Trees and Shrubs ≥6 m Tall

Density of trees and shrubs ≥6 m tall showed a possible age-related trend, however planting density may have biased these results. The time-series curve (Figure 5B) was suggestive of an age-related pattern, with median annual density maintaining a constant low level from year 6 through year 8, then increasing in years 10-11. Results of statistical analysis were weakly significant (Kruskal-Wallis, directional: $p = 0.0561$). This pattern might be expected due to the length of time required for individuals to achieve 6 m in height. However, years 6 through 8 were planted at higher densities than

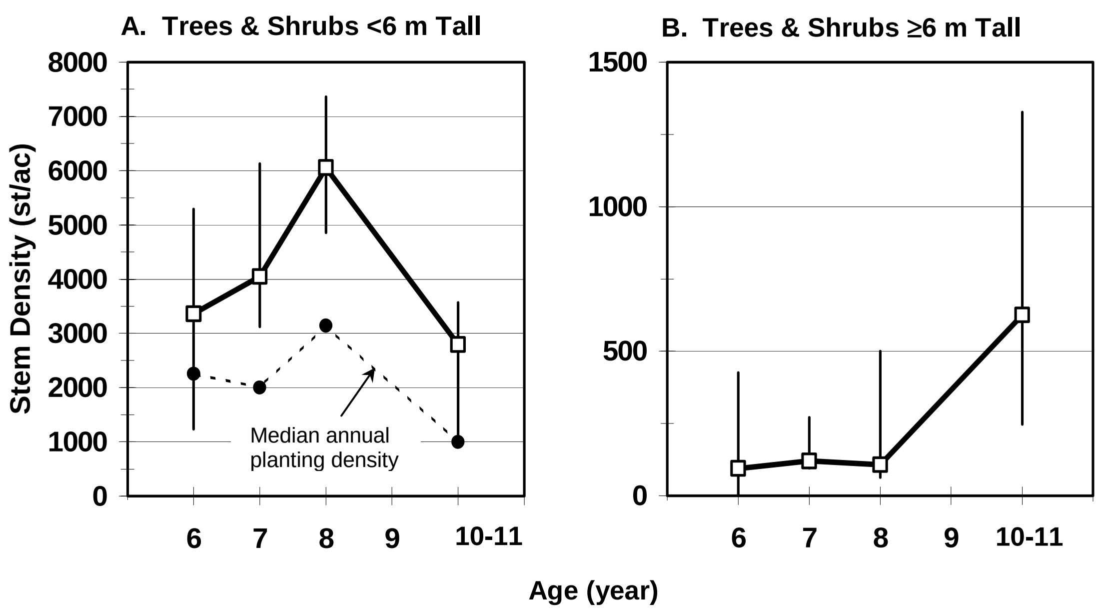

Figure 5. Time-series curves for current stem density (st/ac) of trees and shrubs A) <6 m tall, and B) ≥6 m tall. Boxes (□) indicate the median of each age group; vertical lines represent the interquartile range. Darkened circles (●) indicate the median planting density (st/ac) of each age group.

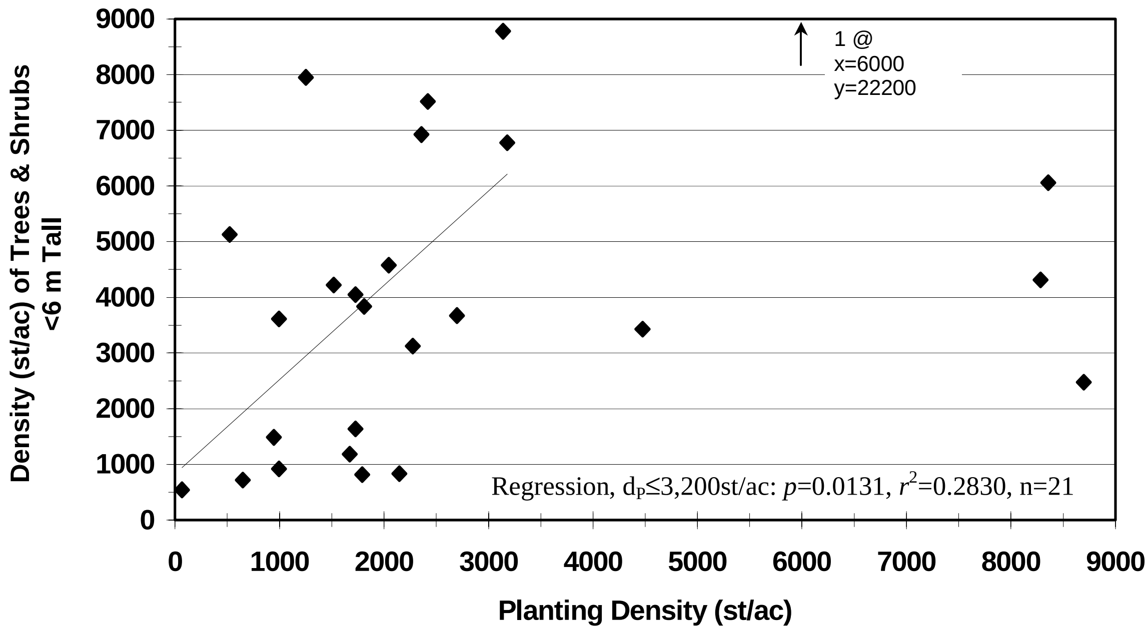

Figure 6. Current stem density (st/ac) of trees and shrubs <6 m tall versus planting density (st/ac). The line indicates results of simple linear regression for planting densities ≤3,200 st/ac (3.7 ft oc).

years 10-11, which may have left the former more susceptible to stand stagnation and repressed tree heights than the latter. The extent of this effect could not be determined.

3.2.3 Combined Stem Density

Density of all woody plants was related to and showed moderate to substantial increases over planting density. Current density showed a significant relationship with planting density for RU's planted at ≤3,200 st/ac (linear regression: $p = 0.0126$, $r^2 = 0.2853$, n = 21). This was similar to the relationship observed between planting density and density of trees and shrubs <6 m tall, since this layer comprised a substantial majority of the total.

Density generally showed moderate to considerable increases from planting to present (Figure 7). Current density was greater than planting density by a median factor of 2.3 for RU's planted at ≤3,200 st/ac. Most RU's in this range increased by factors of 1.1 to 3.1, presumably through natural recruitment and reproduction of planted individuals. The magnitude of increase from planting to present generally appeared to diminish with increasing planting density, however this was not verified statistically.

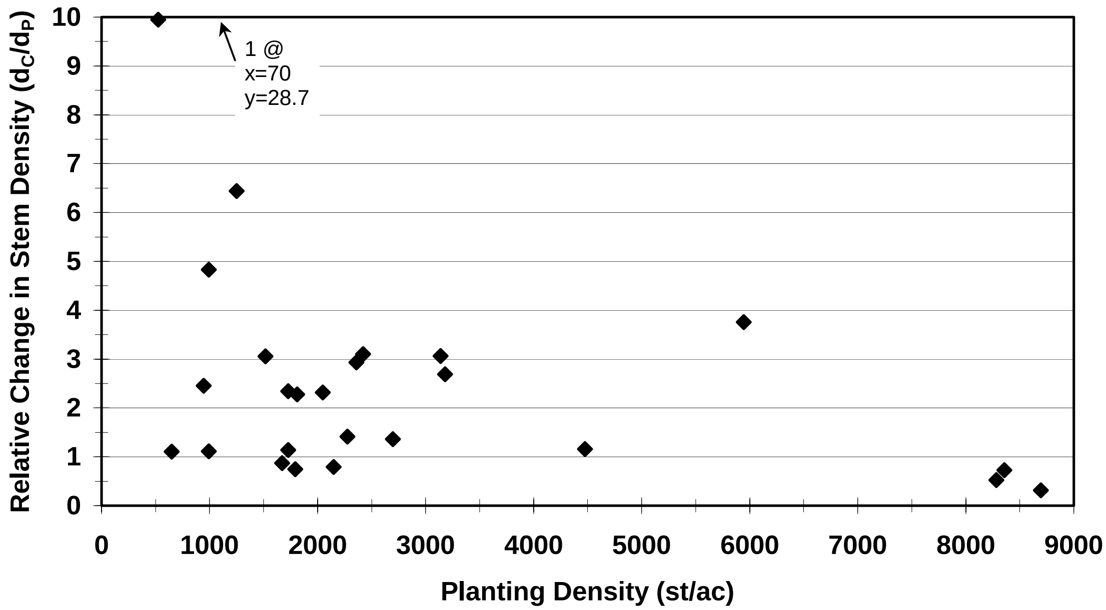

Figure 7. Relative change in stem density of trees and shrubs from planting to present. Relative change is represented by the product of current density divided by planting density.

3.3 Abundance of Woody Nonnative Invasive Species

Aerial cover of nonnative woody vegetation showed no apparent change with age (Figure 8), and maintained a yearly median ≤5%. Statistical analysis confirmed that all age groups were equivalent (Kruskal-Wallis, nondirectional: $p = 0.2062$). When all RU's were considered together, median cover was 3%, with most sites ranging between 0% and 5% cover.

3.4 Abundance of Reed Canarygrass

Reed canarygrass cover was strongly associated with density of trees and shrubs <6 m tall, and was not related to site age or to canopy cover. The time-series plot was suggestive of a possible age-related pattern, with cover values apparently declining from year 6 through year 8, and increasing in years 10-11 (Figure 9). However, this trend mirrored that of density of trees and shrubs <6 m tall (Figure 5A). Further analysis found that reed canarygrass cover was strongly related to density of trees and shrubs <6 m tall regardless of site age: greater densities of the <6 m tall layer were associated with less reed canarygrass (linear regression: $p = 0.0025$; $r^2 = 0.3015$, n = 28) (Figure 10). Since density of trees and shrubs <6 m tall was related to planting density and not site age, the apparent pattern was likely not age-related.

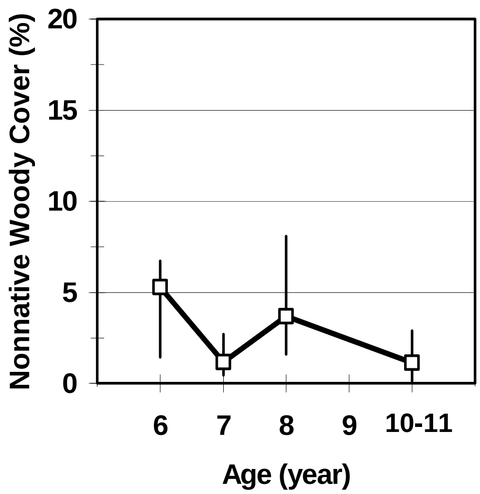

Figure 8. Time-series curve for aerial cover (%) of nonnative woody vegetation. Boxes (□) indicate the median of each age group; vertical lines represent the interquartile range.

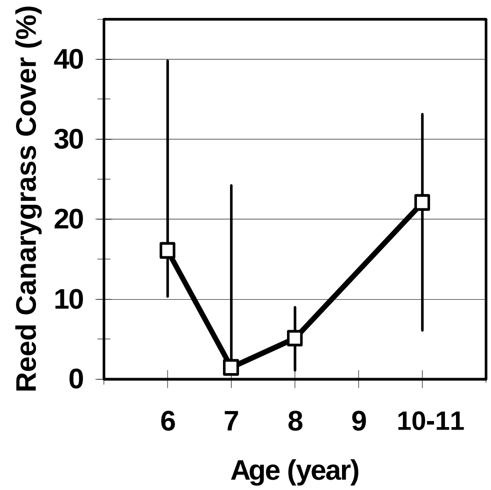

Figure 9. Time-series curve for aerial cover (%) of reed canarygrass. Boxes (□) indicate the median of each age group; vertical lines represent the interquartile range.

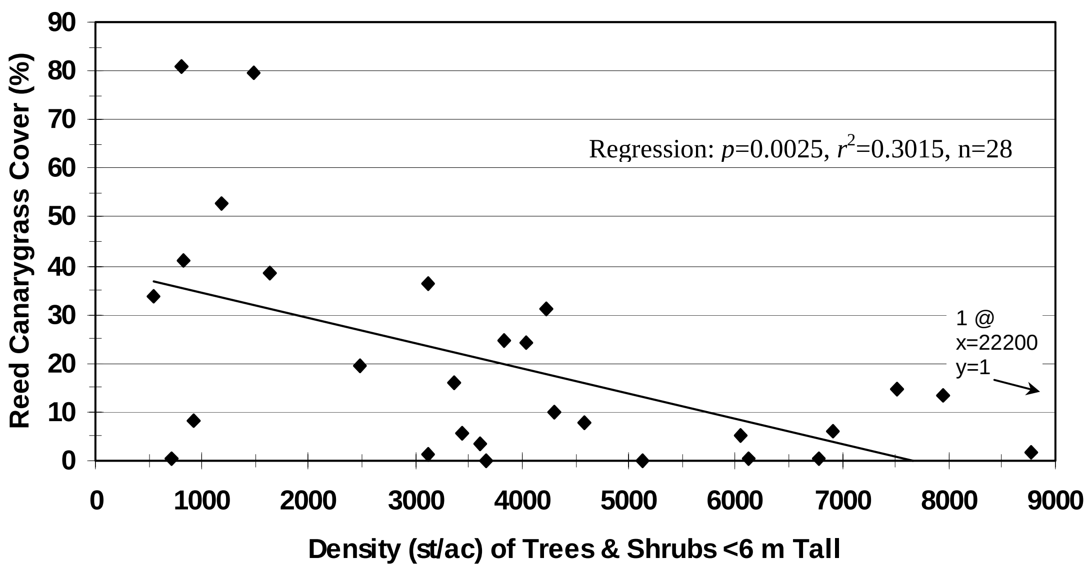

Figure 10. Aerial cover (%) of reed canarygrass versus stem density (st/ac) of trees and shrubs <6 m tall. The line indicates results of simple linear regression, excluding the datum at (x=22200, y=1).

Reed canarygrass cover values often exceeded 10%, and were highly variable across age classes as well as within each age class (Figure 9). When all RU's were considered together, median reed canarygrass cover was 10%, with most sites ranging between 2% and 31% cover. Reed canarygrass cover was not related to canopy cover (linear regression: $p = 0.8168$; $r^2 = 0.0021$, n = 28) (Figure 11).

There were some indications that reed canarygrass cover was related to planting density, however this relationship could not be confirmed. Since reed canarygrass cover was strongly related to density of trees and shrubs <6 m tall, and this density was related to planting density, it follows that reed canarygrass cover would be related to planting density. However, a direct analysis of reed canarygrass cover versus planting density failed to detect a significant relationship (linear regression: $p = 0.2028$; $r^2 = 0.0667$, n = 26) (Figure 12). Research units planted at high densities (>~3,000 st/ac) did appear to have substantially less reed canarygrass, however these were not well represented (n = 7).

3.5 Other Stand Attributes

A complete analysis of all stand attributes was not performed for this study. However, some basic indicators were drawn from the data in order to provide a general sense of plant community structure and dynamics. These features included overall species richness and establishment of planted and volunteer species.

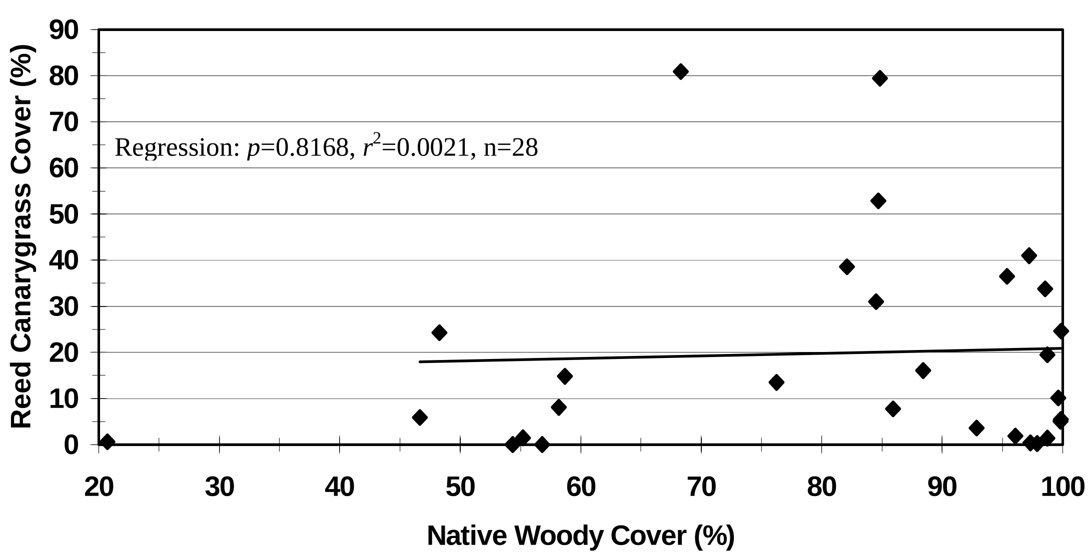

Figure 11. Aerial cover (%) of reed canarygrass versus aerial cover (%) of native woody vegetation. The line indicates results of simple linear regression, excluding the datum at (x=21, y=1).

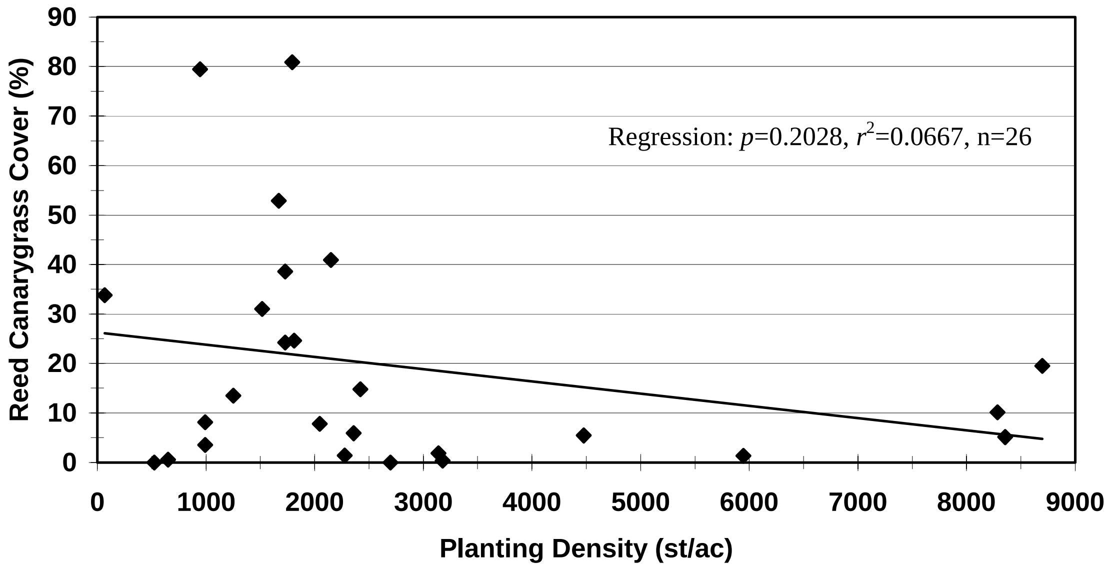

Figure 12. Aerial cover (%) of reed canarygrass versus planting density (st/ac) of trees and shrubs. The line indicates results of simple linear regression.

3.5.1 Richness of Woody Species

In general, woody species richness increased from planting to present, apparently due to successful establishment of planted species and widespread recruitment of volunteer species. As might be expected, the most common and abundant species appeared to be native, early-successional and wetland-adapted.

A total of 16 tree species and 26 shrub species were represented in the data (Table 3). The most abundant in terms of frequency and cover appeared to be red alder (*Alnus rubra*), willows (*Salix* spp.), black cottonwood (*Populus balsamifera*), and red osier dogwood (*Cornus sericea*), and to a lesser extent Oregon ash (*Fraxinus latifolia*), salmonberry (*Rubus spectabilis*), and Pacific ninebark (*Physocarpus capitatus*).

Woody species richness ranged from 5 to 24 species per RU, with a median of 10 species (Figure 13). Most RU's ranged between 8 and 12 species, with 6 to 8 species showing mean cover values ≥1%, and 2 to 3 at ≥10%. Species richness had a strong positive relationship with the number of species planted, such that more species planted resulted in greater species richness (linear regression: $p = 0.0056$, $r^2 = 0.2780$, n = 26) (Figure 14). Species richness did not appear to be a function of time.

Table 3. Woody species encountered during surveys, including frequency of occurrence and median cover within RU's where observed.

Species[a,b]	Occurrence (no. of RU's)	Median cover (%)	Species[a,b]	Occurrence (no. of RU's)	Median cover (%)
Tree Species			**Shrub Species**		
Alnus rubra	25	33	*Cornus sericea*	25	11
Salix lucida var. *lasiandra*	24	17	*Rubus armeniacus*[c]	21	3
Salix sitchensis	22	21	*Rubus spectabilis*	21	1
Fraxinus latifolia	16	1	*Spiraea douglasii*	18	2
Populus balsamifera var. *trichocarpa*	15	3	*Physocarpus capitatus*	11	6
Thuja plicata	14	1	*Rosa nutkana*	11	3
Salix scouleriana	7	2	*Sambucus racemosa*	8	1
Picea sitchensis	5	<1	*Rubus laciniatus*	7	1
Populus tremuloides	3	28	*Symphoricarpos albus*	7	<1
Prunus emarginata	3	2	*Lonicera involucrata*	6	5
Salix species	2	90	*Rubus parviflorus*	6	3
Pinus contorta var. *contorta*	2	5	*Crataegus douglasii*	6	1
Frangula purshiana	2	4	*Cytisus scoparius*	5	1
Tsuga heterophylla	2	1	*Oemleria cerasiformis*	5	1
Pseudotsuga menziesii	1	3	*Corylus cornuta*	4	6
Acer macrophyllum	1	<1	*Rosa* species	4	1
Amelanchier alnifolia	1	<1	*Malus fusca*	3	<1
			Rubus ursinus	2	5
			Acer circinatum	2	4
			Ilex aquifolium	2	<1
Unidentified Tree or Shrub Species			*Viburnum edule*	1	8
Unidentified	5	4	*Philadelphus lewisii*	1	1
			Ribes sanguineum	1	1
			Rubus leucodermis	1	<1
			Ribes species	1	<1
			Vaccinium ovatum	1	<1

[a] Nomenclature follows the United States Department of Agriculture (USDA) PLANTS database (USDA, NRCS 2002).
[b] Common names are provided in Appendix C.
[c] This species is incorrectly termed *R. procerus* in the PLANTS database (USDA, NRCS 2002). A brief discussion of various scientific names commonly used for this species, as well as the correct name used here, is provided by Ceska (1999).

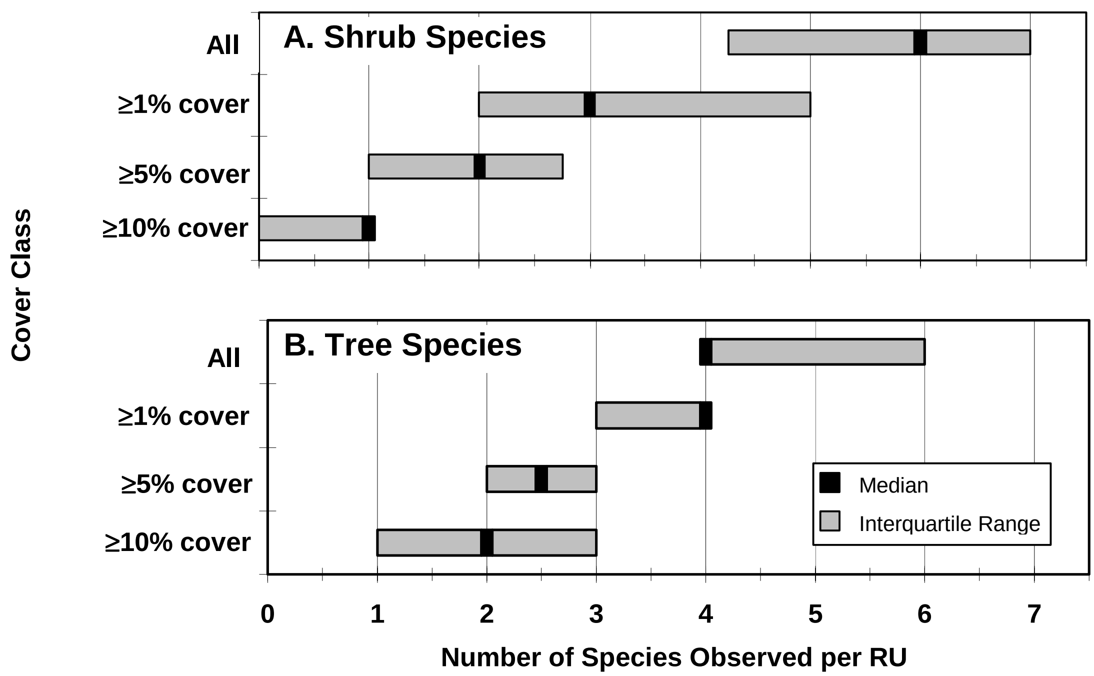

Figure 13. General levels of species richness observed for A) shrub species, and B) tree species. Species richness is represented by the median and interquartile range of all RU's surveyed. Richness is provided for species showing various levels of cover, including all species present and species present at ≥1%, ≥5%, and ≥10% cover.

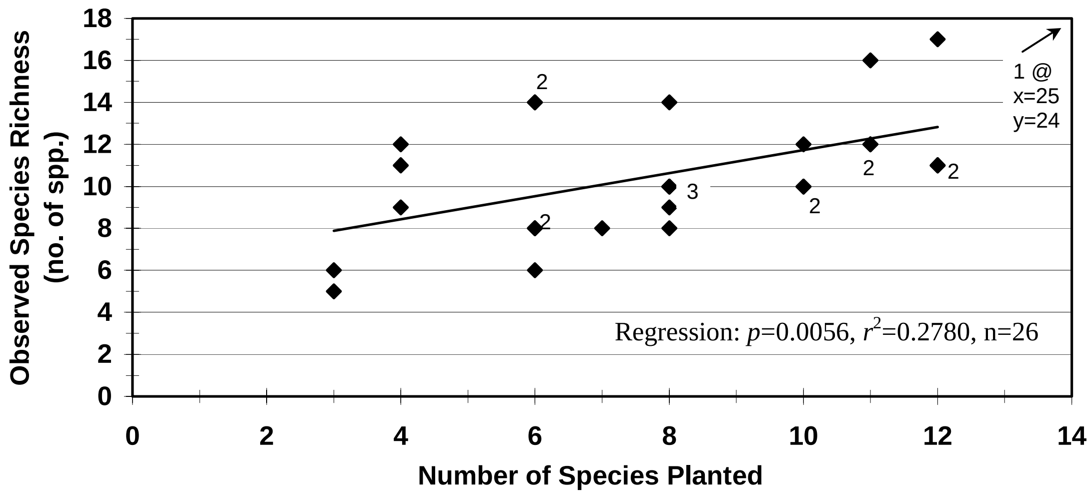

Figure 14. Current species richness versus the number of species planted. The line indicates results of simple linear regression, excluding the datum at (x=25, y=24). Numbers next to data points indicate the number of data represented at the adjacent point, if more than one.

3.5.2 Planted Species Establishment

In general, most planted species were observed during site visits (Figure 15). Early-successional, wetland-adapted species appeared to show the most success (Table 4). Less tolerant species more adapted to uplands or associated with later successional stages also appeared to establish well, albeit at lower levels of cover.

One or more planted species failed to establish at 11 of 27[11] RU's, while 16 of 27 RU's retained all species that were planted. The number of failed species appeared to increase with increasing numbers of planted species (linear regression: $p = 0.0042$, $r^2 = 0.2947$, n = 26). However, as noted above, greater numbers of planted species also resulted in greater species richness. In addition, failure rate was generally low, with ≤2 species lost in 6 of the 11 RU's that lost species. There did not appear to be any relationship between species failures and unit age.

3.5.3 Volunteer Species Richness and Cover

Volunteer species augmented richness and cover at most RU's. Volunteer species were present at all RU's and provided a substantial source of cover for many (Figure 16). Volunteer species provided 3-44% relative cover at most RU's. Over half of all RU's had volunteer species contributing ≥10% of total cover, while one-quarter had volunteers contributing ≥50%.

[11] Two RU's were excluded due to uncertainty regarding planted species.

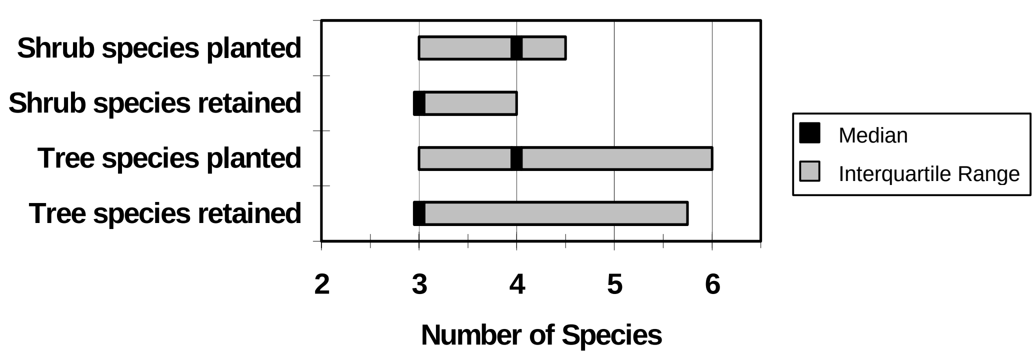

Figure 15. Retention of planted tree and shrub species. The number of species planted and retained are represented by the median and interquartile range of all RU's surveyed.

Table 4. Establishment of planted species. The number of RU's in which each species was planted (N_p) and retained (N_r) is shown. Research units where species volunteered were not included. The establishment rate (R_E) was calculated as the percentage of species planted that were also observed ($[N_p/ N_r]*100$). Median cover of established species (M_{cE}) is also provided.

Species planted[a]	N_p	N_r	R_E (%)	M_{cE} (%)	Species planted[a]	N_p	N_r	R_E (%)	M_{cE} (%)
Tree Species					**Shrub Species**				
Salix lucida					*Cornus sericea*	26	24	92	8
var. *lasiandra*	18	18	100	17	*Rubus spectabilis*	14	14	100	1
Salix sitchensis	17	17	100	26	*Physocarpus capitatus*	12	10	83	6
Thuja plicata	17	11	65	1	*Rosa nutkana*	10	10	100	2
Populus balsamifera					*Rubus parviflorus*	6	4	67	10
var. *trichocarpa*	15	10	67	5	*Lonicera involucrata*	5	5	100	6
Alnus rubra	14	14	100	23	*Symphoricarpos albus*	5	4	80	<1
Fraxinus latifolia	11	10	91	3	*Sambucus racemosa*	4	4	100	3
Picea sitchensis	5	5	100	<1	*Crataegus douglasii*	4	3	75	2
Tsuga heterophylla	4	2	50	1	*Oemleria cerasiformis*	4	2	50	1
Populus tremuloides	3	3	100	28	*Rosa* species	3	3	100	unk[b]
Salix scouleriana	3	3	100	3	*Spiraea douglasii*	3	3	100	6
Pinus contorta					*Acer circinatum*	2	2	100	4
var. *contorta*	3	2	67	5	*Corylus cornuta*	2	2	100	6
Prunus emarginata	2	2	100	2	*Viburnum edule*	2	1	50	8
Pseudotsuga menziesii	2	2	100	4	*Philadelphus lewisii*	1	1	100	1
Salix species	2	2	100	90	*Ribes sanguineum*	1	1	100	1
Acer macrophyllum	1	0	0	-	*Vaccinium ovatum*	1	1	100	<1
Amelanchier alnifolia	1	0	0	-	*Holodiscus discolor*	1	0	0	-
Betula papyrifera	1	0	0	-	*Malus fusca*	1	0	0	-
Prunus virginiana	1	0	0	-					

[a] Nomenclature follows the USDA PLANTS database (USDA, NRCS 2002).
[b] Median cover could not be calculated for this species.

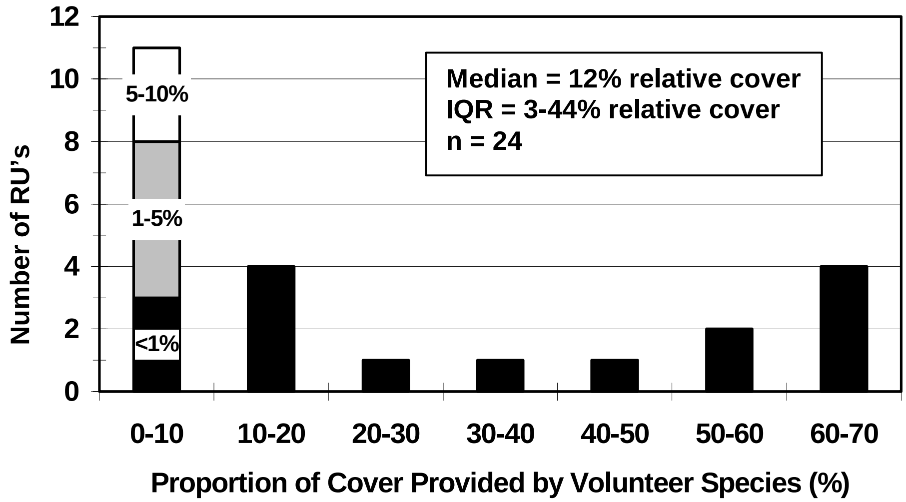

Figure 16. Relative cover provided by volunteer species. Proportions were calculated as the percentage of woody cover (cumulative for all species) provided by volunteer species (cumulative cover of volunteer species).

Volunteer species contributed 2-3 shrub species and 0-2 tree species within most RU's (Figure 17). These species generally consisted of native wetland trees and nonwetland[12] shrubs (Table 5). Nonnative species were frequently encountered, usually at low levels of cover. Red alder (*Alnus rubra*) was the most prominent volunteer species, occurring on 11 of the 15 units where it was not planted, and showing 48% median cover where it volunteered. The most frequently encountered volunteer species was Himalayan blackberry (*Rubus procerus*), although this species showed relatively low cover. The most common native volunteer was Douglas spiraea (*Spiraea douglasii*), appearing at 14 of the 26 units where it was not planted.

[12] *Wetland* species are defined as those with an indicator status of FAC, FACW or OBL, as indicated in the draft revision of the 1996 National List (Reed 1996). *Nonwetland* species are defined as those with an indicator status of FAC-, FACU or UPL.

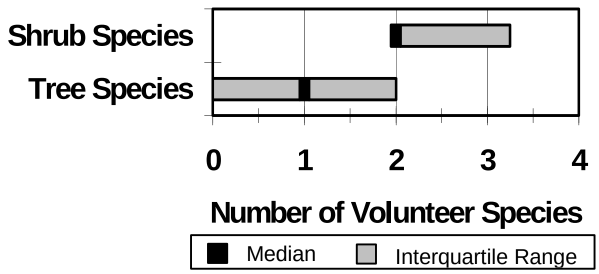

Figure 17. Number of volunteer tree and shrub species observed per RU. Number of species is represented by the median and interquartile range of all RU's surveyed.

Table 5. Establishment of volunteer species. The number of RU's in which each species volunteered (N_v), and the median cover provided (M_v) are shown.

Species[a]	N_v	M_v (%)	Species[a]	N_v	M_v (%)
Tree Species			**Shrub Species**		
Alnus rubra	11	48	*Rubus procerus*	22	3
Salix lucida var. *lasiandra*	5	17	*Spiraea douglasii*	14	1
Fraxinus latifolia	5	<1	*Rubus laciniatus*	8	<1
Populus balsamifera var. *trichocarpa*	4	<1	*Rubus spectabilis*	6	1
Salix sitchensis	3	4	*Cytisus scoparius*	5	<1
Salix scouleriana	3	1	*Malus fusca*	3	unk[b]
Thuja plicata	2	1	*Crataegus douglasii*	3	<1
Acer macrophyllum	1	<1	*Rubus parviflorus*	2	8
Amelanchier alnifolia	1	<1	*Rubus ursinus*	2	5
Prunus emarginata	1	unk[a]	*Sambucus racemosa*	2	1
			Symphoricarpos albus	2	<1
			Corylus cornuta	2	<1
			Oemleria cerasiformis	2	<1
			Rosa species	1	1
			Rubus leucodermis	1	<1
			Lonicera involucrata	1	<1
			Ilex aquifolium	1	<1
			Ribes species	1	<1

[a] Nomenclature follows the USDA PLANTS database (USDA, NRCS 2002).
[b] Median cover could not be calculated for this species.

4.0 DISCUSSION

4.1 General

The results of this study represent characteristics of successfully established stands of native woody vegetation within a broad diversity of mitigation types, designs, vegetative communities, and ecological conditions. Despite this diversity, several overarching trends and relationships were detected. These relationships were generally coarse, yet the ability to detect them underscores their strength and commonality.

By focusing on successfully established stands of woody vegetation, the findings of this study are believed to represent reasonably achievable results of common mitigation practices. This study thus provides data that may help meet the need identified by Ossinger (1999) for "data-based guidelines" in order to "…increase the validity of success standards [used] to determine if [a] mitigation site is developing and functioning as expected." This should not be misinterpreted as meaning that the attributes assessed in this study are in themselves sufficient to assess wetland functions. Rather, these attributes are just a few of the several indicators often used in determining the level of functioning present (Azous et al. 1998; Hruby et al. 1999; Cooke 2000; Null et al. 2000).

This is not to suggest that the results of this study can or should be universally applied. The wide variability evident in the data underscores the need for caution in strictly applying the findings. Furthermore, despite embodying a variety of mitigation conditions, some features were more represented than others in this study. These included:

- vegetative communities - most RU's showed red alder and/or willow species as dominant or co-dominant;
- hydrogeomorphology - 21 of 29 RU's, or 72%, exhibited depressional hydrogeomorphology;
- size - 23 of 29 RU's, or 79%, were less than 1 acre in size; and,
- location - 15 of 29 RU's, or 52%, were located in King county.

Results of this study may thus be more applicable to projects conforming more closely to these features. For example, one site surveyed was planted almost exclusively with Oregon ash at a relatively low density. Given the low planting density and slower growing nature of the species, this site would be less likely to achieve high levels of cover quickly. Indeed, it showed only 21% cover at year 10. However, this site was not intended to rapidly attain high cover, and actually met its success standards.

4.2 Time-Series Curves

The time-series curve for aerial cover of native woody species was the only one believed to provide a valid representation of age-related change. Other time-series curves either showed no evidence of age-related change, or were believed to substantially violate the assumption of accurate representation. The effect of violating this assumption was evident in the influence variations in planting density exerted over time-series curves for density of trees and shrubs <6 m tall, reed canarygrass cover, and perhaps also density of trees and shrubs ≥6 m tall. These curves were suggestive of possible age-related trends, yet were shown to be substantially influenced by design parameters (i.e., planting density) that appeared to differ between age classes.

Native woody cover may have also been influenced by planting density; however, this attribute was likely more resilient to bias. This is based on the failure of planting density to show a direct connection with native woody cover. In addition, planting density showed the greatest difference between years where cover was nearly identical (years 8 and 10-11). Cover appeared to vary more during years where planting densities were more similar (years 6, 7, and 8). Thus, although planting density likely had some influence on native woody cover, the general shape of the time-series curve and suggested timing of critical events (e.g., attainment of 80% cover) are believed to be sufficiently approximated.

In addition to planting density, other features such as HGM class and mitigation type also appeared disproportionately represented within age groups. The effects of these could not be assessed due to insufficient data. These differences may not have translated into substantial bias, however. This is because these classification systems may not accurately represent the actual ecological variables that drive stand development, such as soil conditions and hydroperiod.

4.3 Benchmark Standards and Management Implications

4.3.1 Aerial Cover of Native Woody Species

Results indicate that year 8 is an appropriate benchmark standard for attaining 80% aerial cover of native woody species. Year 7 was actually the earliest to show a median ≥80% (M = 85%) (Figure 1), thereby meeting the criteria established in Section 2.1.2 for identifying a benchmark standard. However, substantial uncertainty arose over the reliability of using this figure as a benchmark due to the relatively small sample size (n = 5) and large variability (IQR = 55-98%) of this age group. In contrast, the year 8 median was well in excess of 80% (M = 96%), and was represented by a larger sample size (n = 7) and smaller variability (IQR = 81-98%). Year 8 may thus provide a conservatively long time-frame for achieving the 80% threshold; however, the uncertainties associated with the year 7 data make year 8 a more reliable target.

Several management implications regarding rapid establishment of high cover levels were evident in the findings. It should be noted, however, that rapid establishment of high cover levels may occur at the expense of other desirable attributes. For example, canopy cover is known to converge earlier when greater numbers of stems are present (Oliver and Larson 1996). This may imply that managers should plant at high densities in order to establish a full canopy as quickly as possible. However, greater stem densities are also believed to contribute to stand stagnation and repressed growth, resulting in shorter individuals with smaller diameters than a lower density stand (Oliver and Larson 1996). This could delay the development of other desirable attributes, such as plant maturity, emergence of a forested canopy, and vertical stratification.

Problems associated with high stem densities and stand stagnation may be offset with appropriate management actions, such as mechanical thinning. Thus, a resource manager may elect to plant at a high density in order to achieve rapid establishment of high cover levels. Then, when the canopy nears convergence, mechanical thinning could be performed to allow the stand to develop without stagnating. This may be conducive to both rapid establishment of canopy cover as well as unimpeded development of a more mature stand.

Management actions such as thinning have not traditionally been used in wetland mitigation. Although there appears to be some ecological benefit to incorporating such actions, the costs of implementation may be prohibitive and were not evaluated as part of this study. Until the costs and benefits of various management actions are further explored, wetland resource managers should weigh long-term consequences against short-term benefits of their planting methods.

This study found two related attributes that appeared pertinent to rapid canopy cover establishment. First, maximum levels of cover (≥90%) were found when trees and shrubs that were ≥2 m tall occurred at densities ≥2,100 st/ac (4.6 ft oc) (Figure 2). This may thus represent a minimum post-planting survival density of species that grow taller than 2 m *if high levels of cover are desired early and effects of stand stagnation are not a concern*. Lower densities would likely result in delayed canopy convergence and less stagnation, while higher densities may result in earlier canopy convergence and greater stagnation.

In addition, planting densities of ≥3,000 st/ac (3.8 ft oc) appeared to more reliably result in high canopy cover (Figure 3), although this could not be confirmed and should therefore be considered tentative. This figure does seem to correspond with the 2,100 st/ac (4.6 ft oc) tall plant density discussed above: Figures for planting density are for all species regardless of how tall they grow, and they do not include post-planting mortality.

4.3.2 Stem Density of Trees and Shrubs

Benchmark standards were not proposed for stem density of trees and shrubs. This is because: 1) stem density is not generally used as a measure of success in wetland mitigation; and, 2) results were insufficient to offer any sort of quantitative guidance.

Although density is not generally used to assess mitigation success, managers may benefit from tracking density and identifying critical stages of stand development. For example, managers wishing to develop a dense understory of desirable vegetation may be unable to do so until the stand nears the end of stem exclusion. This is because stem exclusion often leaves a sparsely vegetated understory, as discussed in Section 1.3.1. Thus, once stem exclusion nears completion, desirable understory vegetation may be more successfully established. Planting understory vegetation earlier may leave the plantings subject to stem exclusion mortality. Planting later or not planting at all may delay understory development and may allow colonization of the understory by invasives or other undesirable species. Further discussion on the latter in relation to reed canarygrass is provided in Section 4.3.4 below.

This study was unable to detect any general age-related patterns in density despite indications that substantial and widespread changes were actually occurring. Such indications included: 1) the general increase in density observed from planting to present; and, 2) physical evidence of stem exclusion observed at many older sites[13]. Studies that track individual sites through time would be more sensitive to detecting these changes.

The general increase observed in stem density from planting to present suggests that volunteering and reproduction can be, and generally are, important sources of woody vegetation at mitigation sites. It is interesting to note that even after often dramatic increases in density, density during years 6-11 was still found to be dependent on planting density. This underscores the role of sufficient planting density in "jump starting" stand development.

4.3.3 Abundance of Woody Nonnative Invasive Species

Results indicate that no more than 5% cover of nonnative woody species is an appropriate benchmark standard during years 6-11. Median cover of nonnative woody vegetation fluctuated between 1-5% during this time (Figure 8). In general, nonnative woody vegetation does not appear to be a problem in wetlands where native woody vegetation has been successfully established. This may be due to: 1) effective control during the monitoring period combined with resistance of established stands to infestation; and/or, 2) because many nonnative species are less adapted to thrive and dominate in wetlands.

4.3.4 Abundance of Reed Canarygrass

Results were not sufficient to establish a reliable benchmark for reed canarygrass. During years 6-11, reed canarygrass cover fluctuated between 1-22% median aerial cover (Figure 9), but showed no age-related trend. In addition, differences in reed canarygrass cover were not associated with woody canopy cover (Figure 11), but were associated

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[13] Evidence for stem exclusion consisted of numerous and randomly scattered dead trunks shorter in length than the existent canopy, and/or a relatively open and unvegetated understory. Some elements of the data also suggested a relationship between canopy cover, density of trees and shrubs ≥6 m tall, and density of trees and shrubs <6 m tall; however, the data were insufficient to confirm such relationships.

with differences in density of trees and shrubs <6m tall (Figure 10). It was not certain how much reed canarygrass existed at the end of year 5, and how much developed after year 5. None of the stands were believed to be in active management for reed canarygrass. These factors prohibited the establishment of a clear and reliable benchmark standard for reed canarygrass during years 6-11.

That reed canarygrass cover was related to density of trees and shrubs <6 m tall, but not with canopy cover has two important implications. First, facilitating a dense initial layer of trees and shrubs <6 m tall may help prevent severe reed canarygrass infestations early in the life of the site. Second, the common perception that canopy cover effectively “shades out” reed canarygrass may be due more to the understory-clearing effects of the stem exclusion stage than to an inability of reed canarygrass to exist under a canopy. A possible corollary is that stands in later stages of development may be subject to re-infestation if a higher density understory is not present. For example, a stand transitioning from stem exclusion to understory reinitiation is likely to have a sparsely vegetated understory that could be subject to reed canarygrass colonization.

Qualitative observations do suggest, however, that reed canarygrass does not form the dense monocultures in the understory that are characteristic of more open areas. Nonetheless, the findings of this study show that reed canarygrass can exist at relatively high levels (as much as 40%) under abundant canopy cover (≥95%). This study was not able to determine: 1) whether reed canarygrass was actually spreading through the understory, or whether it was a remnant from earlier stages; or, 2) the extent to which reed canarygrass inhibits establishment of desirable plant species during understory reinitiation. These relationship should be further explored in order to develop a fuller understanding and contribute to more informed management.

Findings indicate that densities ≥4,000 st/ac (3.3 ft oc) help maintain reed canarygrass below 20% aerial cover (Figure 10). In addition, planting density may be a factor in helping to minimize reed canarygrass, although this relationship could not be confirmed. Preliminary indications from this study suggest that a minimum planting density of 3,000 st/ac (3.8 ft oc) may be optimal. This could not be confirmed, however, and should therefore be considered tentative.

4.3.5 Richness of Woody Species (Trees and Shrubs Combined)

The recommended benchmark standard for richness of all woody species is the number of planted species, up to a total of 12 species. This is based on the following observations: 1) current richness depended on the number of species planted (Figure 14); 2) all RU’s studied, except for one, were planted with ≤12 species; and, 3) richness almost always increased from planting to present, with the magnitude of increase diminishing at higher planting numbers (i.e., RU’s planted with more species showed little increase in richness, while RU’s planted with fewer species showed comparatively larger increases in richness).

Despite the general increase in species richness observed from planting to present, results indicate that stands experienced medians losses of 1 shrub species and 1 tree species per stand (Figure 15). Two factors appeared to offset these losses. First, volunteer species from off-site provided median gains of 2 shrub species and 1 tree species per stand (Figure 17). These often contributed substantially to stand canopy cover (Section 3.5.3; Figure 16). Additional gains likely occurred by species spreading from one area of a site into another, such as buffer plants moving into the wetland. This was observed on many sites, but the extent of it was not evaluated. Given these dynamics, managers and regulators should expect species composition to change somewhat from that planted while species richness increases or remains the same.

4.3.6 Richness of Tree and Shrub Species

The recommended benchmark standards for richness of tree species and shrub species richness separately are 4 tree species and 6 shrub species. These figures represent median measures for all RU's combined (Figure 13), and thus do not incorporate differences in planted species richness, or differences in the number of trees vs. shrubs planted. This has two important implications: 1) higher numbers are probable when relatively large numbers of species are planted; and, 2) they may be inadequate for scrub-shrub wetlands where few if any tree species are planted. These recommended benchmarks may thus represent conservatively low targets when a mix of tree and shrub species is desired.

4.3.7 Richness of Dominant Species

This study did not seek to define or evaluate species dominance. However, some of the findings may be useful to managers seeking to establish a certain number of species at more than just minimal levels. Thus, the following benchmark standards are offered:

- 4 tree species and 3 shrub species at ≥1% aerial cover/species
- 2 tree species and 2 shrub species at ≥5% aerial cover/species
- 2 tree species and 1 shrub species at ≥10% aerial cover/species

These figures represent median measures for all RU's combined (Figure 13), and thus do not incorporate differences in planted species richness, or differences in the number of trees vs. shrubs planted. This has similar implications as those discussed in Section 4.3.5.2 above. These recommended benchmarks may thus represent conservatively low targets when a mix of tree and shrub species is desired.

5.0 SUMMARY AND RECOMMENDATIONS

This study successfully documented basic features and relationships of woody plant stands at mitigation projects between 6-11 years of age. Recommended benchmark standards and other relevant findings are summarized in Table 6. These results represent a variety of mitigation types, designs, vegetative communities, and ecological conditions within western Washington lowland mitigation sites. Due to this diversity, results of the study should be applied with care and considerable site-to-site variability should be expected.

The method used to construct time-series curves is believed to have adequately served the main purpose of the study, which was to identify broadly applicable general trends and relationships useful for establishing benchmark standards. More precise results might be obtained by controlling for influential factors such as planting density and hydroperiod, among others. Other methods, such as tracking individual sites over time, are likely more sensitive to detecting trends that may have been obscured in this study.

In consideration of these finding, the following recommendations are offered:

1. Use the benchmarks identified in Table 6 to establish success standards for new projects, keeping in mind the noted considerations. This is not to suggest that all benchmarks listed should be employed on every project, or that only attributes for which benchmarks are provided should be used to evaluate success. Rather, success standards for mitigation projects should be based on the functions desired, and attributes chosen to evaluate success should reflect these functions. The benchmarks offered by this study merely provide a set of reasonable and achievable structural attributes. It is left to mitigation managers and regulators to decide which if any should be used to evaluate specific wetland functions and mitigation success.

2. Periodically evaluate and update the proposed benchmark standards as advancements in mitigation science and practice warrant.

3. Further study relationships between the development of desired stand attributes. Identify benchmark standards that are optimum for attaining multiple desired attributes. For example, explore impact of rapid canopy cover establishment on development of other potentially desirable features, such as plant maturity, emergence of a forested canopy, and vertical stratification. Develop standards that do not favor one desirable attribute to the detriment of others.

4. Further study the influence of planting density on canopy expansion and reed canarygrass dynamics. As a starting point for future study, consider 3,000 st/ac (3.8 ft oc) as a tentative minimum planting density that contributes to rapid establishment of woody cover and suppression of reed canarygrass.

5. Consider maintaining a high density shrub layer (i.e., trees and shrubs <6 m tall) in order to help suppress reed canarygrass, assuming the impact of such densities on other desirable attributes is acceptable. Shrub layer densities ≥4,000 st/ac (3.3 ft oc) appear to limit reed canarygrass abundance to <20% aerial cover. This appears effective either when the shrub layer forms the uppermost stratum, or when it is in the understory of a forest canopy. The latter may require supplemental plantings of shade tolerant understory shrubs and trees when sites emerge from stem exclusion into understory reinitiation.

6. Incorporate concepts of forest stand dynamics (i.e., stand initiation, stem exclusion and understory reinitiation) into management of forested and scrub-shrub wetland mitigation zones. Consider tracking these stages at mitigation sites and making management decisions appropriate to the stage.

7. When rapid establishment of canopy cover is desired, consider 2,100 st/ac (4.6 ft oc) as a minimum post-installation survival density for species expected to grow ≥2 m tall. This density of tall plants appears critical to canopy convergence during years 6-11.

Table 6. Proposed benchmark standards, with considerations for implementation and other relevant findings.

Proposed benchmarks	Considerations for implementation	Other relevant findings
AERIAL COVER OF NATIVE WOODY SPECIES		
80% aerial cover by year 8	• may not be conducive to development of other potentially desirable attributes (e.g., plant maturity, emergence of a forested canopy, and vertical stratification)	• generally increases until year 8 • high levels require ≥2,100 st/ac of tall (≥2 m) plants • planting densities ≥3,000 st/ac may be optimum for canopy convergence during years 6-11, but this could not be confirmed
STEM DENSITY OF WOODY SPECIES		
none proposed		• substantial increase from planting to present (1.1-3.1x for planting densities ≤3,200 st/ac) • density during years 6-11 depends in part on planting density • age-related change implied but not confirmed
ABUNDANCE OF WOODY NONNATIVE INVASIVE SPECIES		
≤5% aerial cover during years 6-11		• no age-related change indicated • established stands of native species appear capable of maintaining low levels during years 6-11 without management intervention
ABUNDANCE OF REED CANARYGRASS		
none proposed		• variable, often high levels (M = 1-22% during years 6-11) • no age-related change indicated • changes with shrub layer density, not canopy cover • planting densities ≥3,000 st/ac may be optimum for maintaining minimal levels during years 6-11, but this could not be confirmed
RICHNESS OF WOODY SPECIES		
number of species planted	• valid for up to 12 planted species • conservatively low	• no age-related change evident • species composition may change slightly
RICHNESS OF TREE AND SHRUB SPECIES		
4 tree & 6 shrub species/stand	• did not consider differences in planted species richness, or differences in number of trees vs. shrubs planted • probably conservatively low • higher numbers likely with > numbers of planted species	
RICHNESS OF DOMINANT WOODY SPECIES		
4 tree & 3 shrub spp. @ ≥1% aerial cover/spp. 2 tree & 2 shrub spp. @ ≥5% aerial cover/spp. 2 tree & 1 shrub spp. @ ≥10% aerial cover/spp.	• did not consider differences in planted species richness, or differences in number of trees vs. shrubs planted	

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APPENDIX A

Appendix A. Study site and research unit information by site.

Study Site Information								Research Unit Information		
Site	Age (year)	County	Mitigator	Type[a]	Adjacent Land Use[b]	WSDOE Rating[c]	HGM Class[d]	RU ID (if >1/site)	Size (ac)	Cowardin class[e]
BGC	6	Clark	WSDOT	C	R	2	D-O		0.45	SS
BGE	6	Clark	WSDOT	C	R	2	D-C		0.25	SS
BGW	6	Clark	WSDOT	C	R	3	D-C		0.32	SS
BNG	8	King	Private	R	GB	3	D-C		0.42	FO
BSO	7	King	WSDOT	C	D	3	D-C		0.55	FO
BUR	8	Kitsap	WSDOT	C	N	2	D-C		1.98	FO
CED	7	King	WSDOT	C	D	3	D-O		0.13	SS
CEN	10	Lewis	Public	R	D	3	R-I		1.81	FO
CIC	7	Snohomish	WSDOT	E	R	2	D-O		0.50	FO
COR	10	Whatcom	Private	E	D	2	D-O		0.28	FO
CRD	10	Kitsap	Public	R	D	3	D-O		0.22	SS
JCR	7	Kitsap	WSDOT	R	R	2	S		0.24	FO
KEN	10	King	Public	C	R	2	D-C	1	0.14	FO
								2	1.15	SS
LAC	7	King	Private	E	D	3	D-C		0.55	FO
MAN	6	King	WSDOT	E	R	3	R-F		0.71	FO
MAS	6	King	WSDOT	E	R	3	R-F	1	1.13	FO
								2	0.41	SS
								3	2.59	SS
MEB	10	King	WSDOT	C	GB	3	D-O		0.09	SS
MEI	8	King	WSDOT	E	GB	2	D-O	1	0.18	SS
								2	0.35	SS
NIS	11	Pierce	Public	C	GB	2	D-O		1.03	FO
SAM	10	King	Private	R	GB	2	D-O		0.12	FO
SIP	8	Snohomish	WSDOT	C	D	3	D-C		0.43	FO
SIS	8	Snohomish	WSDOT	C	D	3	S	1	0.17	FO
								2	0.12	FO
SUG	11	King	Private	R	D	3	D-O		0.49	FO
TRY	10	Kitsap	Public	R	D	3	D-C		0.21	SS

a C = creation; R = restoration; E = enhancement.
b D = developed; GB = greenbelt; R = rural; N = natural. See text for definitions.
c WSDOE 1993.
d Hruby et al. 1999. D-C = depressional closed; D-O = depressional outflow; R-F = riverine flow-through; R-I = riverine impounding; S = slope.
e Cowardin et al. 1979. FO = palustrine forested; SS = palustrine scrub-shrub.

APPENDIX B

Appendix B. Mean values of native and nonnative woody cover, woody plant stem density, reed canarygrass and estimated planting density of each RU.

RU	Aerial Cover Data (%) Native Woody	Nonnative Woody	Reed Canarygrass	Existing stem density (st/ac) for given height class <1 m	1-1.9 m	2-2.9 m	≥6 m	Planting Density (st/ac)
BGC	59	3	15	2420	3680	1410	0	2420
BGE	57	0	0	540	2940	200	0	2700
BGW	47	13	6	3680	2800	430	0	2360
BNG	54	3	0	3220	1070	840	90	520
BSO	48	<1	24	390	1070	2590	0	1730
BUR	76	<1	13	1980	2340	3620	110	1250
CED	99	0	1	1150	1370	19690	120	5950
CEN	21	0	1	60	160	500	0	650
CIC	85	1	53	500	140	540	270	1670
COR	85	5	31	1890	1090	1250	420	1520
CRD	100	1	25	850	560	2430	290	1810
JCR	98	5	<1	1990	1600	2540	1830	0
KEN-1	93	3	4	310	770	2540	1180	990
KEN-2	58	0	8	100	200	620	190	990
LAC	55	3	1	1320	600	1200	100	2280
MAN	88	6	16	810	1330	1220	100	unk[a]
MAS-1	82	8	39	170	450	1020	340	1730
MAS-2	97	5	41	40	10	780	870	2150
MAS-3	68	0	81	90	120	600	520	1790
MEB	100	3	5	1000	810	1620	1760	4480
MEI-1	100	4	10	340	220	3760	0	8280
MEI-2	100	1	5	1320	1480	3260	40	8360
NIS	85	2	79	80	980	420	840	950
SAM	95	3	37	450	1370	1290	2150	unk[a]
SIP	86	9	8	1610	1330	1640	160	2050
SIS-1	96	28	2	2140	2260	4380	850	3140
SIS-2	97	8	<1	1540	2900	2340	1770	3180
SUG	99	1	34	110	150	280	1380	70
TRY	99	0	20	320	300	1860	230	8700

[a] Panting density could not be determined due to incomplete historical documentation.

APPENDIX C

Appendix C. Scientific names, common names, codes, and nativity status for species encountered. Nomenclature is from USDA PLANTS database (USDA, NRCS 2002)

Scientific name	Common name	Code	Nativity[a]
	Tree Species		
Acer macrophyllum	big-leaf maple	ACMA	N
Alnus rubra	red alder	ALRU	N
Amelanchier alnifolia	Saskatoon serviceberry	AMAL	N
Betula papyrifera	paper birch	BEPA	N
Frangula purshiana	cascara	FRPU	N
Fraxinus latifolia	Oregon ash	FRLA	N
Picea sitchensis	Sitka spruce	PISI	N
Pinus contorta var. *contorta*	shore pine	PICO	N
Populus balsamifera var. *trichocarpa*	black cottonwood	POBA	N
Populus tremuloides	quaking aspen	POTR	N
Prunus emarginata	bitter cherry	PREM	N
Prunus virginiana	choke cherry	PRVI	N
Pseudotsuga menziesii	Douglas-fir	PSME	N
Salix lucida var. *lasiandra*	Pacific willow	SALU	N
Salix scouleriana	Scouler willow	SASC	N
Salix sitchensis	Sitka willow	SASI	N
Salix species	willows	SALI	N
Thuja plicata	western red cedar	THPL	N
Tsuga heterophylla	western hemlock	TSHE	N
	Shrub Species		
Acer circinatum	vine maple	ACCI	N
Cornus sericea	red osier dogwood	COSE	N
Corylus cornuta	beaked hazelnut	COCO	N
Crataegus douglasii	Douglas' hawthorne	CRDO	N
Cytisus scoparius	Scotch broom	CYSC	I
Holodiscus discolor	oceanspray	HODI	N
Ilex aquifolium	holly	ILAQ	I
Lonicera involucrata	black twinberry	LOIN	N
Malus fusca	western crabapple	MAFU	N
Oemleria cerasiformis	Indian plum	OECE	N
Philadelphus lewisii	mock-orange	PHLE	N
Physocarpus capitatus	Pacific ninebark	PHCA	N
Ribes sanguineum	red-flowering currant	RISA	N
Ribes species	currants & gooseberries	RIBE	N
Rosa nutkana	Nootka rose	RONU	N
Rosa species	roses	ROSA	N
Rubus armeniacus[b]	Himalayan blackberry	RUAR	I
Rubus laciniatus	evergreen blackberry	RULA	I
Rubus leucodermis	black raspberry	RULE	N
Rubus parviflorus	thimbleberry	RUPA	N
Rubus spectabilis	salmonberry	RUSP	N
Rubus ursinus	trailing blackberry	RUUR	N
Sambucus racemosa	red elderberry	SARA	N
Spiraea douglasii	Douglas spirea	SPDO	N
Symphoricarpos albus	common snowberry	SYAL	N
Vaccinium ovatum	evergreen huckleberry	VAOV	N
Viburnum edule	highbush-cranberry	VIED	N

[a] N = native; I = nonnative invasive

[b] This species is incorrectly termed *R. procerus* in the PLANTS database (USDA, NRCS 2002). A brief discussion of various scientific names commonly used for this species, as well as the correct name used here, is provided by Ceska (1999).