

AN ABSTRACT OF THE THESIS OF

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Title: Applying Economic Benefit Transfer to Improve the Transfer of Ecological Estimates in Ecosystem Services Research & Policy

Abstract approved:

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The goal of this thesis is to advance the methodology and thought regarding the transferability of ecological estimates of ecosystem services. Conceptually and in practice, ecological estimate transfer parallels economic benefit transfer in ecosystem services research and policy, yet the literature for benefit transfer predates ecological estimate transfer by several decades. The economic benefit transfer literature has identified context similarity to be a major prerequisite for the accurate transfer of economic estimates. This thesis applies approaches and lessons learned from economic benefit transfer to develop a framework for conducting ecological estimate transfers and a structure for describing ecological contexts.

Despite the need and utility of ecological estimate transfers for data-limited ecosystem services research and policy there lacks a consistent method for evaluating such transfers across contexts. This thesis proposes a framework for describing and evaluating contextual variables in order to add consistency and rigor to transferability practices. Such guidance is needed in order to assess the need for more sophisticated treatment of uncertainty and error. The relationships between structural or ecological elements (i.e., context) and specific ecological processes (i.e., the data generating processes, or production functions) may be numerous and complex, however, broadly the nature and existence of ecological processes may be described in terms of its scale and location, termed here the ‘contextual reference frame’. The contextual

reference frame is proposed in this thesis as a structure for ecological contexts and basis for transferability assessment to identify and explore sources of transfer error.

Assumptions and challenges inherent in transferability assessment are illustrated in a case study of benthic microalgal primary production (BMPP) estimates, which reflects data and parameter transfers used in fisheries-habitat ecosystem service assessments. The case study applies the framework approach to qualitatively and quantitatively explore contextual variables and determine a basis to transfer estimates from the literature to a hypothetical policy site. The case study highlights the relative utility of simple univariate (ANOVA) and more complex multivariate classification methods (CART) analysis.

This thesis finds that the proposed framework facilitates the representation of the major sources of error and uncertainty associated with transfers. Benefit transfer practices have increased the awareness and exploration of transferability limitations in policy and research applications. Use of the framework can promote analogous awareness and future research may increase the consistency and reliability of ecological estimate transfers. The case study finds that future research should focus on replicating transferability tests across contextual levels and variables to develop better indicators of transfer reliability and define acceptable limits of transfer error and uncertainty.

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes the release of my thesis to any reader upon request.

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GLOSSARY OF TERMS

Ecosystem Services: The benefits people obtain from ecosystems, as both goods and services that are tangible and intangible (Millennium Ecosystem Assessment 2005).

Benefit Transfer: “The [application] of existing estimates of nonmarket values to a new study which is different from the study for which the values were originally estimated” (Boyle & Bergstrom, 1992).

Ecological Estimate Transfer: The application of existing estimates of ecological structures or processes to a new study which is different from the study for which the estimates were originally estimated.

Data Generating Process (DGP): The mechanism through which estimates (either economic or ecological) are produced in reality. Models and functions are representations of the ‘true’ DGP.

Meta-Analysis: The synthesis, analysis and summary of a collection of studies (Osenberg et al. 1999).

Scale: The space over which processes are observed and defined by. Indexed by both extent and grain. “The spatial scale of ecological data encompasses both grain and extent” (Wiens 1989; Turner et al. 1989).

Extent: “the overall size of the study area. For example, maps of 100 km² and 100,000km² differ in extent by a factor of 1000.” (Turner et al. 1989)

Grain: “the resolution of the data, i.e., the area represented by each data unit. For example, a fine-grain map might organize information into 1-ha units, whereas a map with an order of magnitude coarser resolution would have information organized into 10-ha units.” (Turner et al. 1989)

Contextual Reference Frame: Structure proposed by this thesis to describe the context of a study or estimate in terms of its spatial, temporal and ecological organizational scale in terms of both extent and grain of scale. The contextual reference frame also refers to the point location of the study or observation which can be described spatially, temporally and ecologically as well.

Context: The space within which investigations are conducted and to conclusions apply. Includes the reference frame and other factors and variables which affect an estimate’s value.

Extrinsic contextual variable: Study or site features that may not be useful for describing an ecological or economic process within a site but may be useful for describing differences in process across sites (e.g., factors or features that are constant across original sites such as geographic setting or habitat type).

Intrinsic contextual variable: Theoretically or empirically important core driving variables for a given ecological or economic process.

Scaling: transfer of information *across* scale levels i.e., interpolation and extrapolation.

Transfer error: Difference between the transferred estimate and the ‘true’ estimate for a given site where the ‘true’ estimate may be known (as in validation tests) or unknown. Major contributions to this error are measurement error, publication bias and generalization errors.

1. Introduction

There is increasing demand to describe and account for the values and uses that humans obtain from ecosystems in decision-making and management (Pearce et al. 1989; Daily et al., 2009; EPA, 2009). However, comprehensive descriptions of these benefits, referred to as ecosystem services, and their production can be limited either because some services and their connections to human benefit are less well understood (e.g., biodiversity), or because particular services have not been studied for the contexts (e.g., location, ecosystem type, or scale) for which assessments are desired. In particular, quantifying multiple ecosystem services for a given context is often difficult due to the quantity of multidisciplinary information that is required. In part, this creates a demand to directly apply existing estimates of ecological production, even though the original context (i.e., the study site) of the estimates often differs from where the estimate is needed (i.e., the policy site). Additionally, management relevant ecosystem service information may seek to incorporate information concerning the ecological production of ecosystem service endpoints in order to trace changes in ecosystem structure (such as habitat conversion), species composition or ecological processes to ecological endpoints that humans use and value. Such ecological production functions, which describe the quantitative relationships between goods and services and the inputs required to produce them (Barbier 2007) are important for assessing trade-off and scenarios (Granek et al. 2010).

While methodologies exist to analyze the transfer of economic ecosystem service values (e.g., benefit transfer), there is no analogous formalized approach for the transfer of ecological data for estimating ecosystem service production. As a result, data may be transferred with little to no identification or analysis of uncertainties associated with differences in study and policy site contexts, or comparisons to other sources of error. Ecological estimate transfer represents a

useful tool for research, policy and management to gain ecosystem service information when primary research is not available, however guidance is needed for assessing assumptions, trade-offs, and the need for primary research. This thesis works to apply foundational principles from benefit transfer, in particular the role of context similarity, for evaluating transfer errors in ecological estimate transfers.

1.1 Examples of Ecological Estimate Transfer for Ecosystem Service Production

One of the most notable examples of economic ecosystem service value transfer is from Costanza et al. (1997). However, less recognized is that this study also transferred *ecological* estimates. In this paper, worldwide economic values were estimated. In their estimation of services worldwide by biome, ecological estimate transfers across space and scale were utilized. For example, food production as an ecosystem service was calculated for estuaries by relating primary production rates of these systems to the fish production rate in these systems, and by applying an average price to obtain the ecosystem service value in dollars per year (see supplementary information for Costanza et al. 1997). However, the average primary production rate used in this calculation comes from ten U.S. East and South-East coastal estuaries (Houde and Rutherford 1993), and is applied as an average to the areal extent of estuaries globally.

Examples of ecological transfers can be found across the ecosystem services literature. It may be difficult to map or model multiple ecosystem services production on a landscape due to the lack of sufficient biophysical information in the desired location. For example, Egoh et al. (2008) utilized previous model results, specifically the median annual runoff in order to map multiple ecosystem services production in South Africa. It may also be desirable to express the importance of particular ecosystem services using a synthesis of available data. For example,

Grabowski et al. (2012) valued services provided by oyster reefs. However, Grabowski et al. (2012) use an average estimate from a biogeographic region (southeastern United States) in their calculations and apply to oyster reefs generally. Kroeger and McMurray (2008) survey ecosystem services provided by Yaquina Bay, Oregon. In order to estimate carbon sequestration rates in saltmarsh ecosystems, the authors synthesize estimates from across the literature. The estimates reported included a recent average estimate from four California saltmarshes, a 100-year average from San Francisco South Bay, and an average rate of tidal marshes in the conterminous U.S. The authors used the highest estimate (the U.S. average) and the lowest estimate (the California average) to construct high and low scenarios.

Across these studies, assumptions regarding the applicability and accuracy of transfers for the intended purpose are inconsistently discussed and investigated. For example, studies which aggregate estimates as an average may use a similar habitat or ecosystem type as the stated filter, or basis for transfer. With respect to these studies it can be unclear what the definition of the ecosystem type is (e.g., a particular classification scheme), and whether this is a valid approach for selecting estimates. Explicit comparison of transferred values in terms of other sources of accuracy such as the method used or the variability around the estimate may also be unstated. In addition, the effect of variability (real or simulated) around a chosen mean estimate on the final assessment or model results may also not be acknowledged.

While many different goods and services' production can be described in terms of transferred ecological information, this thesis focuses on a particular service, the importance of aquatic and coastal habitats for fisheries production (habitat-fisheries services). This service describes the value of habitats for fishery support as refugium, food, or nursery support (Barbier et al 2011).

1.2 Need for Transferability Assessment: Evidence from GecoServ

The Gulf of Mexico Ecosystem Service Valuation Database was queried (GecoServ, Plantier-Santos, Carollo, and Yoskowitz 2012) to identify studies that estimated the value of coastal habitats to fisheries. Studies from this database were utilized for a few reasons including that the database records information on the value, service, valuation method, country of study, and ecosystem type. This was particularly useful since it was hypothesized that values identified as elicited using the production function approach (alternatively, production method) would be more likely to use or link ecological production functions to economic values, and thereby rely on ecological information of ecosystem services production.

Thirty-five valuations of habitat value were exported from GecoServ in September 2013, from 26 studies of several ecosystem types including saltmarshes, combined freshwater and saltwater marshes, and seagrass meadows (21, 7 and 7 valuations respectively). Potential ecological estimate transfers were investigated if a study noted the use of information from another study or from the literature and were recorded as transfers if information was from a different context, usually a different location, scale or ecosystem. Ecological estimate transfers were utilized in 10 of 26 papers and underly 9 of 19 reported estimates that utilized either a production method or benefit transfer approach. In total, this represents 12 of the total 35 estimates in the database for the reviewed ecosystem types. Therefore, over one-third of the economic valuations surveyed in this small study relied upon transferred ecological data, increasing to nearly one half of studies when considering only those which utilized the production method or a benefit transfer approach. However, the validity of any given transfer was not assessed in this initial survey.

The observed prevalence of ecological estimate transfers represents an important issue especially given the stated importance of the ecosystem service production method, or other approaches that are able to relate environmental changes to changes in human benefit (Barbier 2007; Daily et al. 2009). Parameter estimates of primary production rate or natural mortality rate were commonly extracted from the literature. However, justifications regarding the suitability of transferred estimates for the given study varied. For example, some researchers prioritized estimates from proximate geographic locations (e.g., Feagin et al. 2010), or aggregated estimates across similar ecosystem types (e.g., Johnston et al. 2002). Some explicitly compared the range of estimates chosen for transfer (e.g., from a similar geographic range) to other potential estimates from a different contextual scope (e.g., from a global range), to illustrate differences and potential context dependencies (e.g., McArthur and Boland 2006; Kroeger and McMurray 2008), however many others did not present such a comparison or validation for the chosen transfer approach (e.g., Watson, Coles and Lee Long 1993). Therefore, it appears that often *a priori* assumptions concerning the potential similarity or applicability of estimates for ecosystem service models are treated inconsistently in the available literature. While there may be logical or theoretical reasons to expect ecological estimates to be more similar if closer together in space, or collected in similar ecosystems, the relationship and interaction of these features may be important. Is geographic proximity or ecosystem similarity sufficient for valid transfers? For example, for a policy site in California is an estimate from California 40 years ago more reliable than a more recent estimate from similar ecosystems in Oregon? In addition, is a single site transfer between a very similar policy and study site more reliable than an average value transfer based on several more dissimilar study sites? Such questions are worth investigating, in order to

understand the effect of transfer assumptions and to foster the utility of ecosystem service research for policy applications.

As has been done for economic estimates, this work hopes to bring attention to ecological estimate transfers and related sources of error. In addition, this work suggests a potential method to facilitate the exploration and acknowledgement of ecological estimate transfer limitations. Drawing from key concepts of benefit transfer, this thesis presents a framework by which the accuracy of ecological estimate transfers may be assessed based on comparisons of the ecological contexts at study and policy sites, and on comprehensive evaluation of assumptions, sources of error, and uncertainty. This framework is then used to analyze the transferability and of benthic microalgal primary productivity rates to hypothetical policy site using a large number of estimates from coastal sites globally. While not specifically ecosystem service production estimates per se, benthic primary production and other primary production estimates may be used as parameters in habitat-based, coastal fisheries production studies and models (Barbier et al. 2011; Costanza, Farber, and Maxwell 1989; Johnston et al. 2002; McArthur and Boland 2006). This analysis serves to illustrate the potential utility of the framework and associated graphical and statistical methods to evaluate the transferability of ecological data, such as those useful estimating for ecosystem service production.

2. Benefit Transfer: Conceptual Foundation for Ecological Estimate Transfer

The need for time and cost effective surveys and assessment of ecosystem service values as part of mandated cost-benefit analyses has prompted the study of economic value transfers between sites (Bergstrom and De Civita 1999). While the use of benefit transfer was common as early as the 1980s, it wasn't until the early 1990s that researchers began to formalize procedures

and protocols (Johnston and Rosenberger 2010). At present, this literature spans the theoretical implications and repercussions for validity and precision resulting from this practice to guiding frameworks developed for academics, policy-makers and managers to help assist decision-making about how to perform transfers (Ranganathan et al 2008). In part, this has resulted in widespread recognition and acknowledgement of the term ‘benefit transfer’ in the literature, so that those who utilize it in their research commonly self-identify the practice (Brenner et al. 2010; Camacho-Valdez et al. 2013; Kubiszewski et al. 2013).

2.1 Benefit Transfer Framework for Transferability

As previously mentioned, there is extensive guidance for benefit transfer inside and out of the academic literature (Boyle and Bergstrom 1992; Desvousges, Johnson and Banzhaf 1998; Bergstrom and De Civita 1999; EPA 2000; Loomis and Rosenberger 2006; Navrud and Ready 2007). A framework that presents major components and considerations for conducting benefit transfer comes from within an EPA report for preparing economic analyses (EPA 2000). This framework serves as a compelling basis to define an analogous framework for the transfer of ecological estimates given its components are applicable overall. Addressing uncertainty in terms of contextual similarity in addition to other sources is a major component of the EPA framework. It also notes the importance of reporting and assessing assumptions:

“Benefit transfer involves judgments and assumptions. Throughout the analysis, the researcher should clearly describe all judgments and assumptions and their potential impact on final estimates, as well as any other sources of uncertainty inherent in the analysis.” (EPA 2000, p.87)

In order to support the application and re-interpretation of a benefit transfer-based framework, next major sources of error in benefit transfers will be outlined and compared to errors described for ecological studies.

2.2 Sources of Error and Uncertainty in Transferred Values

The benefit transfer literature has identified that the primary contributions to the total errors associated with transfers of economic value are primary study measurement error, publication bias, and site correspondence error (or generalization error) (Liu et al. 2011; Rosenberger and Stanley 2006).

Measurement error is the variability in the original estimate that arises during data collection due to analytical imprecision and natural random error. This reflects the potential divergences between a true underlying value and the primary study estimate (Rosenberger and Stanley 2006). Measurement error is nearly identically described for ecological analysis. Measurement errors occur during the collection and analysis of ecological data, including random error, instrument bias (for example, thermometers and salinity probes) and methodological choices such as the those concerning the inclusion and adjustment of data (Sugihara and May 1990). This refers to how data are included or excluded from final data used for summary statistics or model building, or if data was adjusted before analysis. For modeled or statistically derived estimates, this also includes variation left unexplained by the model (Guisan, Edwards, and Hastie 2002).

Publication selection bias refers to the nature and value of published estimates that may not be representative of empirical evidence (Rosenberger and Stanley 2006). Publication selection bias results as an artifact of preferentially publishing statistically significant results or

those that are consistent with theoretical expectations. Data may go unpublished because they do not meet study-specific objectives (i.e., data are legitimately pruned by the investigator) or they are not sufficiently novel to meet editorial requirements (i.e., a manuscript is rejected by a journal), yet those data may still be useful for value transfer. These biases have important implications for the validity of estimate distributions (e.g., the completeness of the distributions), therefore affect meta-analyses which depend on the statistical independence and representative distribution of values. This can lead to inaccuracy of a predicted estimate's central tendency or variability. These biases have important implications for the validity of estimate distributions, therefore especially for meta-analyses since it affects the independence and distribution of values, skewing both estimates of central tendency and variability towards a value other than the 'true' value. Such issues are not unique to economics and have been the subject of discussion in ecological disciplines as well (Leimu and Koricheva 2004; Lortie et al. 2007; Nakagawa 2004). That said, publication error is difficult to detect and measure, because the investigator seeking to account for it usually cannot know what is not reported.

Generalization error is defined as uncertainty arising from the transfer process as a result of differences in context between locations where the original measurements were made (hereafter, study site) and the location to which the estimate is to be transferred (hereafter, policy site) (Johnston and Rosenberger 2010). The extent of generalization errors has been quantified in the benefit transfer literature, usually using tests of convergent validity. Tests of convergent validity correspond to validation techniques used in ecology (Rykiel 1996). Generalization errors have been compared across U.S. states (Vandenberg et al. 2001); European countries (Bateman et al. 2011); and various scales (Johnston and Duke 2009). Generally, it has been found that the magnitude of transfer errors decreases as site similarity increases although exceptions have been

noted (Johnston and Rosenberger 2010), which will be discussed later. Therefore, site similarity is often referred to as a fundamental requisite (Johnston 2007; Spash and Vatn 2006). The importance of site similarity for point estimate transfers is emphasized because they often cannot easily be adjusted for differences, as opposed to models with the potential to tailor estimates for policy sites (Boyle et al. 2010; Jiang, Swallow, and McGonagle 2005). That is, models can be adjusted or calibrated for differences in context by using or deriving policy site specific parameters from data that is associated with the policy site.

The previous discussion supports that a benefit transfer approach serves as a useful foundation for an exploration of and framework for ecological estimate transferability. Firstly, foundations and pressures to transfer ecological estimates and economic estimates between studies for use in ecosystem service assessments are similar. Secondly, major sources of error as identified benefit transfers have direct analogs in ecological estimate transfers. Thirdly, benefit transfer theory and methodologies have been distilled into simple, policy-informative guidance frameworks that are useful for researchers and managers, which might be similarly developed to guide ecological estimate transfer.

Because of the key importance of context similarity for the transfer of point economic estimates, this requirement, and generalization error generally, receive focal attention in the development of a framework for ecological estimate transfers. However, defining ecological “sites” or “contexts” requires deeper discussion. In the next section economic contexts will be explored in order to support a definition of ecological contexts.

2.3 Economic Contexts in Benefit Transfer

Lists of conditions for site (context) similarity vary between studies but most include similarity across the following: the environmental good studied, its quantity and the change in quantity/quality; the population; market characteristics; institutional setting; temporal differences between the primary data collection and that of the policy site; and geographical location (Spash and Vatn 2006; Stapler and Johnston 2008; Zandersen, Termansen, and Jensen 2007).

Differences in these requirements have been attributed to the lack of a formal definition of similarity or the elements over which similarity may be measured (Johnston 2007). Using these site-similarity conditions (as listed above) three general requirements are observed: Firstly, similarity in the process or estimate that is needed/transferred (e.g., the environmental good studied, change, quantity and change in quantity and quality, extent of the market); secondly, similarity in core theoretically important drivers (e.g., conceptual and methodological foundations used) and thirdly, similarity across variables that are not necessarily core variables or drivers of the original estimate but have been shown to be useful to explain variation between economic values (e.g., geographic location). These conditions may be further described as ‘intrinsic’ and ‘extrinsic variables’, or variables that are or could be used to describe the production of an estimate at a given site, and variables that could be used to explain differences between sites, respectively.

Both intrinsic and extrinsic contextual variables are important to defining the overall context of a study or policy site. Intrinsic contextual variables are study features that can be used to explain variation in an estimate within a site. These are the direct causal factors that theoretically drive the demand for the good or service, and are typically used as predictor variables in economic models. Extrinsic contextual variables are factors that may be correlated

with the estimate, particularly among study sites, and include general site characteristics (e.g., geographic location or time of measurement) that vary across sites but are relatively constant within a site. That said, extrinsic contextual variables may actually be surrogates (i.e., proxies) for driving factors, or contain driver variables hidden within a composite variable, that are not sufficiently well understood to be included as “intrinsic” variables.

Several assumptions are necessary to describe a site in terms of contextual variables. The theoretical basis that ties both intrinsic and extrinsic characters to the response variable in question (i.e., the “estimate” to be transferred) is the assumption that the response is a function of observable characteristics. Consistency across observable characteristics confers similarity in the mechanism which produces the estimate (the data generating process or DGP) and therefore in the estimate in question (Boyle et al. 2009). Observable characteristics are components and features in a study that can be measured and described. Unobservable characteristics, by contrast, are things that systematically affect the estimate but are not recorded or measured (for example, the health and attitudes of consumers, which is not usually included in economic surveys) (Boyle 2009). In other words, it is expected that the true process that generates the estimate is represented by observable characteristics (Boyle et al. 2009; Kaul et al. 2013), and that those observable characteristics drive the response (i.e., the “estimate”) in the same way functionally at study and policy sites. This is related to explained and unexplained variation in regression models (Kent 1983). Secondly, it is expected that as the values of the intrinsic variables (i.e., those that drive the response) converge between sites, the value of the response (i.e., the “estimate”) will converge. This is represented by tests for statistical equivalence between a transfer model and the true model (i.e., a model specific to the policy site), where if the two models are not identical the transfer is not valid (Downing and Ozuna 1996). For point estimates, this may be practically demonstrated by testing for statistical

equivalence between a transferred estimate and associated model parameters and independently measured values at a policy site (i.e., validation, or tests of convergent validity).

Therefore, ignoring sources of bias and error, two sites with identical context characteristics are assumed to be described by identical functions and produce identical estimates. By extension, differences in observable and unobservable intrinsic context characteristics may result in estimate differences. Observable contextual characteristics at one study location may differ among multiple locations. The ability to describe and account for such differences across studies and sites is a fundamental motivation for the estimation of underlying meta-valuation functions (Rosenberger and Phipps 2007). This tenant thus explains why the comparison of context variables (e.g., intrinsic and extrinsic variables) lies at the heart of economic benefit transfer, and as will be shown later, for the transfer of ecological estimates as well.

2.3.1 “Extrinsic” Contextual Variables

Extrinsic contextual variables are site features that may be unaccounted for or unimportant in a model for a given site, but either directly or indirectly account for differences in the same estimate between different sites. Therefore, extrinsic contextual variables come directly into play when transferring estimates across sites (Rosenberger and Phipps 2007).

“Many of the physical differences important for calibrating values across sites are unmeasured in the original functions. In part, this is because these characteristics are fixed, or constant, in individual site models or researchers assumed these

differences are captured in the price coefficient.” (Rosenberger and Phipps 2007, p. 38)

Extrinsic variables that are constant within a given site may vary across multiple sites. For example, if an estimate was derived over the course of one year, for example 2015, the effect of ‘year’ as a predictor cannot be determined, since there is only one observation, one year, over the course of the study. In that study, ‘year’ is an extrinsic variable. However, if the same estimate was derived over multiple years from multiple studies, the effect of year can be assessed.

Proximity in terms of in-state versus out-of state, domestic versus international, and other physical and geographical characteristics such as topography, lake size or water clarity have all been included in benefit transfer models and in tests of convergent validity (See Rosenberger and Phipps 2007 for a review). Overall, such tests have supported the hypothesis that site similarity is important for transferability. Therefore, it is potentially equally as important to defining the context of an estimate (or the underlying meta-function) to describe driving variables that are unobserved, unaccounted or exogenous to the original functions as it is to identify main theoretical drivers and potential differences in the specification (e.g., parameters) of the original model for a site. Site similarity and DGP similarity are therefore closely coupled, where variables for site similarity may closely follow specifications for economic functions (i.e., consist of core economic variables) or deviate to include other meaningful predictors (such as extrinsic variables).

2.3.2 “Intrinsic” Contextual Variables

Core variables, specific to a process and corresponding estimate (i.e., intrinsic variables) represent a key component of context. Guidance for intrinsic variables represents both economic theory as well as results from empirical investigations, but also variables that are able to be described as either intrinsic or extrinsic. Theoretically guided core variables may more closely represent intrinsic variables, as features that are key to describing the process (In fact, in empirical models these are the core variables measured). Furthermore, core variables have been described as an important requirement for benefit transfer ‘meta-models’ (Bergstrom and Taylor 2006). This framework proposes that in addition to identifying potential extrinsic variables in studies that may vary between a study site and a policy site and contribute to generalization errors, that changes in and variability in ‘core’ process-level variables should also be considered.

This recommendation follows guidance from Bergstrom and Taylor (2006) on formulating meta-analysis benefit transfer (MA-BT) models. Bergstrom and Taylor (2006) discuss how approaches for constructing MA-BT models may strictly follow economic theory (“strong structural utility theoretic”) or may not (“weak” or “non-structural utility theoretic”). Weak structural utility theoretic approaches include deviations from the theoretical model to include other potentially relevant predictors in MA-BT models. An example non-structural utility theoretic meta-analysis of wetland values is provided by Woodward and Wui (2001) where physical and geographic variables such as wetland size, number of services valued and other study characteristics were included but core economic variables were not (e.g., income, substitutes and household characteristics). In this example, it may be observed that such ‘non-structural theoretic’ variables would be classified as extrinsic variables for most studies, since

individual studies are less likely or able to investigate the influence of the size of a given wetland, for example, on the elicited value.

Whether a strong or weak theoretic approach is utilized, Bergstrom and Taylor (2006) discuss the advantages to accuracy and rigor of benefit transfer practices when researchers consider core economic variables and strongly recommend their inclusion in MA-BT models. Therefore, this framework seeks to similarly account for theoretically derived context variables, as intrinsic variables, in addition to extrinsic variables.

So far it has been illustrated that to understand how an estimate changes within and across contexts an understanding of both intrinsic and extrinsic variables is required. Economic contexts have been generally defined by the economic literature as a composite of intrinsic and extrinsic variables. This definition of context was represented by lists of requirements for site similarity, as well as the guidance for defining meta-models which speak to describing important intrinsic and extrinsic variables.

The following framework will illustrate the role of both extrinsic and intrinsic contextual variables for describing and comparing ecological contexts. In addition, the framework outlines the importance of defining the similarity of the estimate or change in an estimate that is desired for transfer. The framework utilizes additional ecological theoretical guidance to define contextual reference frames that may help users select appropriate and theoretically consistent contextual variables. In the following sections first, the framework for conducting ecological estimate transfers will be presented, then further discussion of the basis and current discussion related to the proposed structure of ecological contexts, including reference frames, will be discussed.

3. Framework for Conducting Ecological Estimate Transfers

A framework similar to the benefit transfer guidance provided by EPA (2000) may be helpful for thinking about and ultimately conducting ecological estimate transfers between study and policy sites. Such a framework can provide a process for performing transfers in a consistent manner with explicit acknowledgement of assumptions and potential sources of uncertainty. In this thesis I present a framework to help improve rigor and transparency of ecological estimate transfers in ecosystem services research and policy (figure 1) which builds upon the guidance and theories provided in benefit transfer discussed earlier (table 1). In addition, the framework utilizes additional ecological theoretical guidance to define the scales of investigation, called the contextual reference frame, which may help users better define the estimate for transfer as well as select appropriate and theoretically consistent contextual variables. The characteristics of the ecological estimate framework will be explored in the following sections, and further demonstrated within the case study.

A meta-analytic approach to thinking about transfers across contexts may assist the ability to think about and assess transferability and generalizability assessment in several ways. Firstly, the contextual reference frame of the policy site may be clearly outlined. Secondly, the reference frame of the policy site may be compared to that of the study sites for both correspondence (same or similar reference frame) and coverage (representative). Thirdly, the reference frame may help identify relevant intrinsic and extrinsic contextual variables. Fourthly, gaps in measured variables relative to theoretical constructs may be identified, including defining areas for future research. Defining reference frames provides a coarse baseline for defining the context of both the policy site and the study site/s in a transfer. In addition, inconsistencies across reference frames help identify assumptions which can be more holistically considered and

evaluated. Holistic considerations may be particularly important to ecosystem services research, since it is inherently multidisciplinary and often requires the synthesis of information across disciplines (Wainger and Mazzotta 2011). Therefore, many transfer decisions, such as the choice to transfer estimates that are more similar geographically, or in a similar ecosystem type, reflect researcher knowledge and judgments that may change across disciplines because of differences in standardized or common practices.

This framework (Figure 1) corresponds both with the previously presented framework from EPA's guidance for conducting economic analyses (EPA 2000; Table 1) as well as a related framework on ecological model transferability (Moon et al. 2013; DeWitt et al. 2014). These steps are as follows: First, the context of the application (policy) site and transfer need (estimate) are defined and candidate transfer estimates are found. Next, candidate estimates are individually reviewed for validity (Conceptual Validity). Then, the remaining candidate estimates are compared collectively in terms of their numerical value and context (Operational Validity). Finally, magnitudes of uncertainty are evaluated alongside identified contextual similarities in order to make final transfer decisions.

3.1 Step One: Study Purpose and Data Selection

The first step of an ecological estimate transfer is to determine the nature and purpose of the present study and the desired estimate. What information is needed at the policy site (e.g., ecological parameter or ecosystem service production estimate), in what form (e.g., units) and for what purpose (e.g., policy evaluation, site assessment, hypothesis testing)?

Next is to define the contextual reference frame of the policy (or application) site across categories of scale: space, time and ecological organization. Define the location spatially, temporally, and ecologically (e.g., Oregon, 2015, salt marsh) as well as the scale of investigation across each category of scale in terms of grain and extent. Grain could be defined by assessment/model needs or output (e.g., is the study or model spatially explicit and if so what is the resolution). Extent is represented by the area applicable to the investigation (e.g., Oregon wetlands, or Yaquina Bay estuary). Together, the description of the scale and location of the policy site defines the policy site reference frame. The policy site reference frame description will be useful to identifying and filtering study site estimates that correspond to the policy site. In step three, if study site estimates do not correspond completely, the investigator will have to consider whether to change the expression, extent or resolution of the required estimate to match the available data and with what respect to changes in transfer error and construct validity.

3.1.1 Theoretical and Empirical Guidance

After the policy site reference frame has been defined, relevant contextual variables for the estimate of interest can be determined from literature review. Contextual variables will be used in step three to assess similarity between study and policy sites and include both intrinsic and extrinsic contextual variables. What are the biophysical factors that produce the ecological estimate for the scales (contextual reference frame) of inquiry (i.e., intrinsic variables)? What other context variables are useful for describing the locations where the estimate has been measured and where it will be applied, relevant to the contextual reference frame (i.e., extrinsic variables)?

Intrinsic and extrinsic variable identification begins with the theoretical construct and is guided by a literature review of studies of the estimate of interest, with attention paid specifically to the policy site reference frame. Sources of information to guide literature review may vary, ranging from the predictor variables used in models for the estimate in question, to descriptions of general driving variables covered in meta-analyses. When possible, mechanistic or processed-based equations describing the process that generates the estimate (e.g., for primary production, $[\text{light energy}] + 6\text{CO}_2 + 6\text{H}_2\text{O} \text{ ----} \rightarrow \text{C}_6\text{H}_{12}\text{O}_6 + 6\text{O}_2$) will be useful to elucidate core theoretic variables from extrinsic variables. Both intrinsic and extrinsic variables may include a suite of biophysical and ecological variables or proxies such as species, community type, tidal regime, salinity or light.

Candidate estimates for transfer should be found and pooled from the literature or other data sources based on an initial description of the desired estimate and its context, including important theoretical and empirically important driving contextual variables for the associated process. If direct theoretical or empirical rationales are missing for differentiating the desired estimate based on a perceived driver or contextual variable, strong consideration should be given to omitting the variable in favor of a broader scope, since comparisons and reductions in scope can occur later in transferability assessment. Therefore, considerable literature review occurs at this step, however it may be necessary to revisit the literature or seek best professional judgment for additional guidance at other framework steps. The framework steps are designed to be iterative, where each step can be repeated or reviewed to redefine requirements in order to change the scope of the assessment, provided that the new basis of each step is explicitly stated.

Overall, step one corresponds with the first two steps of the EPA framework (table 1) and also generally reflects the first step of research synthesis, as has been described in the benefit

transfer literature alongside discussion of meta-analysis (Smith and Pattanayak 2002), or the first step of the impact pathways approach for assessing ecosystem services production and value (van Beukering, Cesar, and Janssen 2003).

3.2 Step Two: Conceptual Validity

The step of conceptual validity has been outlined for ecological models (Rykiel 1996) wherein modelers must first determine whether or not the underlying theory and assumptions of a model are valid for an intended use. Similarly, users must evaluate whether estimates, and the procedures used to acquire them, are valid in their own right before judging their suitability for transfer. Rykiel (1996) states that conceptual validity involves stating a scientifically acceptable rationale of the cause and effect relationships in the model, but this logically applies to the process that generates the estimate of interest (the DGP). The related concept of internal validity is used to describe the robustness of scientific experiments (Loewenstein 1999) which also applies to the robustness of integrity of studies that generate ecological estimates.

Both internal and external validity should be given careful consideration in selecting candidate estimates for transfer. Specifically, internal validity is “the ability to draw confident... conclusions from the research” (Schram 2005, p. 226). Considerable attention has been given to the topic of internal versus external validation, or how conclusions from experimental populations can be generalized to real world populations for economic experiments (Levitt and List 2007). External validity is used to describe the ability of experimental contexts and populations to represent real ones for which the experiment is designed to study, or the artificiality of the experimental setting. For example, Moore and Robinson (2004) observe that

the use of artificial bird nests in studies to measure reproductive success reduces the external validity of the study, since predation of artificial nests differs from real nests and reduces the ability for conclusions from studies utilizing artificial nests to be generalized to populations with real nests.

While step two primarily focuses on evaluating the validity of individual estimates for the context that it was originally derived, it may also be important to consider differences in study site estimate contextual reference frames. Differences between reference frames will be a focus of the next step, however at this step differences between reference frames may be considered broadly to ensure that candidate estimates individually are representative of their intended or stated reference frame. Simply stated, this helps to ensure that comparisons across reference frames will be informative since individual estimates are correctly specified, or matched to their reference frame.

As in benefit transfer, the step of conceptual validity may be described as identifying and choosing to include or exclude studies from further assessment based on the rigor or consistency of a particular study with current ecological guidance or theory; i.e., particular methods have been shown to introduce bias, such as a proxy method or a given measurement technique. For example, different methods to measure benthic microalgal production produce estimates that are neither completely representative of net primary production or gross primary production (Cahoon, 1999) so a user may choose to specify a particular technique. An analog in benefit transfer might be choosing to not consider ecosystem service value estimates that were derived using the replacement cost method, since this represents a proxy for the value of the service and relies on assumptions that may not always be valid or based on theoretical constructs of economic value (Brander, Brouwer, and Wagtendonk 2013; Woodward and Wui 2001).

Furthermore, this step corresponds with identifying sources and magnitudes of measurement error and defining limits for acceptable magnitudes. In addition to the previous considerations for measurement error, the amount of variability around an estimate (i.e., as represented by a confidence interval or standard error) also should be considered. This reflects guidance from benefit transfer that measurement error may represent a lower bound on the total amount of variability, or error, around transferred estimates.

“In fact, even if the process of benefit transfer were without error, the transferred value would be expected to differ from the actual value by the square root of the sum of the estimation variances of these two sites.” (Rosenberger and Stanley 2006, p. 374)

If error or variability metrics are not available or reported, users should consider whether to include them in the assessment, since this lower bound of error will not be able to be evaluated in the final steps of transferability assessment.

The basis that any researcher deems a particular approach, method or any other aspect of the scientific process to be defensible is likely to vary between disciplines or the particular field of study, and therefore is subject to professional judgment. In part because ecosystem services research fluidly moves across disciplines, it is integral to this framework that the basis for selecting or excluding estimates for transfer must be made explicit as well as corresponding scientific rationales. If a scientific rationale for exclusion is unclear, or cannot be justified, retaining the estimates will allow for further exploration in the rest of the assessment as long as assumptions are noted for later review.

At this step, justifications should be provided for assumptions and for using simplifications of known processes and for conjectured relationships of poorly known processes

(e.g., derived using indicators or proxies) (Rykiel, 1996). In addition, estimates based on transfer methods should also be earmarked as potentially biased or excluded. For example, careful consideration or further investigation is warranted if estimates are derived from other assessments. It may be found that an estimate derived from another study was in turn based on results from another study, in a chain of transfers. If this is a case, depending on the number of available estimates for transfer, it may be worthwhile to first investigate these transfers retrospectively. Afterwards, a decision can be made if the estimate is valid enough for inclusion in the present transfer.

Overall, the conceptual validity step represents filters that researchers can use to select metadata. This serves the purpose of both evaluating the individual robustness of each estimate or study and for setting minimal requirements for accuracy and error.

3.3 Step Three: Operational Validity and Context Assessment

In step three, candidate estimates are first compared to each other numerically to reconcile differences in definition and in representation, then are compared across contextual variables and reference frames for consistency and coverage (3a). Finally, estimates are compared to the policy site in terms of their numerical value and similarity of the context for which it was estimated, using previously identified contextual variables (3b and 3c). The first part of step three defines the ‘niche’ for which the estimate of interest has been previously measured or calculated across contexts, and then how this relates to the context of the policy site. The application niche allows users to visualize the correspondence of estimates to policy site contextual variable values, to assess representation. Then, users can explore the relationship

between numerical values of the estimate and contextual variables using qualitative or quantitative methods.

Trade-offs exist for decisions made at the previous steps when selecting candidate estimates, specifically for determinations of sufficient accuracy or context similarity. If few estimates are found that meet both requirements, the ability to construct an application niche and explore contextual variable relationships is limited. A user may either conclude that no acceptable transfers exist, or redefine acceptable limits for conceptual validity or context similarity. For the latter, this should only proceed if assumptions and limitations are clearly stated and recorded. Alternatively, requirements may be refined to be more specific if it appears that the bounds are too general or there are many estimates for comparison. Either way, justifications should be provided and scientifically grounded (Boyle et al. 2009).

3.3.1 Evaluate Comparability, Applicability and Representativeness

Estimates are now compared in terms of their definition, numerical value and contextual reference frame.

The EPA guidelines describe commodity consistency as a requirement (i.e., uniformity of the definition of the good that is valued) however theoretical consistency (i.e., the uniformity of the type of value that is estimated) has also been described alongside this requisite (Londoño and Johnston 2012). Together, these relate to the comparability in the definition and representation of the estimates (e.g., net primary production (NPP) versus gross primary production (GPP)). Differences either underlying the definition, the units of representation, or both must be reconciled to compare estimates. In some cases, estimates may be converted to a set of common units; however, extrapolating estimates to new reference frame scales (i.e., upscaling or

downscaling in space, time, or ecological organization) must account for changes in scale-dependency of driving factors that generate the estimate. The framework introduces the concept of the contextual reference frames to draw attention to potential sources of generalization error related to up or downscaling, which will be discussed in detail in the following section.

Similarity across study site reference frames, both among each other and to the policy site reduces additional sources of generalization error and bias and also assists in the identification of relevant intrinsic and extrinsic contextual variables. If candidates do not correspond in terms of their reference frames, it should be investigated if it is reasonable to assume scaling invariance or negligible scaling effects, since scaling effects can affect process-specific intrinsic variables (Gustafson 1998). Therefore, required transformations or adjustments that change study site reference frames should have clear scientific rationales. If, for example, the primary production rate of seagrass beds was measured at sites throughout an estuary and an average estimate across the entire estuary is desired, it is acceptable to calculate an average value if the sampled sites are indeed representative of the seagrass beds across the estuary (see Lussenhop 1974 for history and theory of sampling).

In addition to representing equivalent processes in terms of comparable units, the candidate study site estimates must also have comparable associated contextual variables. Intrinsic and extrinsic variables that were identified during literature review are now associated to study site estimates. These can be specified using either the information that is reported in the original study, or from other sources (e.g., GIS coordinates, specific ecological classifications). If specific biophysical or ecological contextual information is missing from many estimates, users may rely more on readily available extrinsic contextual variables or proxies of intrinsic variables, which limits conclusions in subsequent transferability assessments.

Once a set of valid, site-specific estimates and the associated contextual variables have been compiled, the contextual niche can be explored to ensure representation of policy site characteristics by the selected intrinsic and extrinsic variables. Attributes that are compared in an application niche include the intrinsic and extrinsic variables identified from the literature review in part one. A policy site whose context is outside of the bounds of the application niche (e.g., beyond cumulative geographical representation, ecosystem or habitat type, different depth gradient or salinity and temperature profile) especially for intrinsic variables, may have the highest transfer risk. This is because aspects of the policy site are not represented within the meta-data are therefore “unmeasured”. Unmeasured or unobserved attributes cannot be readily explored in this framework. To construct the application niche, ranges of numerical values and categorical descriptions in the dataset are compared to the policy site in comparable terms. A potentially simple and useful method to construct the application niche is a radar plot, an example of which will be provided in the case study.

3.3.2 Qualitative Context Comparisons

The next step is to explore trends and relationships of the estimate’s numerical value across contextual variables. Numerous graphical methods may be used (e.g., scatterplots, and histograms) to detect both the level of variance or potential trends across individual variables with respect to the response estimate. Consistently identifiable patterns such as directional differences between the desired estimate and a given contextual variable (e.g., the estimate is much lower across polar studies than in tropical studies) may quickly inform decisions concerning transferability, while inconsistencies may confound decision-making at this step. Because this step generally corresponds with approaches for data exploration which is a common step inherent to statistical analyses, statistical method texts (e.g., Ramsey and Schafer 2012) and

references on conducting meta-analysis (see Schulze 2004 for review of meta-analysis methods) can provide additional detail and other methods. Examples of methods potentially useful for transferability assessment will be highlighted in the case study.

3.3.3 Quantitative Context Assessment

Depending on the results of qualitative assessment and policy site needs quantitative analysis may be useful to elucidate clear univariate or multivariate effects or predict a policy site estimate. Similar to qualitative assessment, a suite of approaches may be useful in this step and in particular tools for meta-analysis (see Osenberg et al. 1999; Stewart 2010). Some methods, such as meta-regression may allow for the prediction of a new estimate for the policy site based on its characteristics (Schulze 2004). In addition, quantitative assessment may also allow for much of the final step of transferability assessment, uncertainty analysis, to be performed concurrently. An apparent trade-off, is the time and knowledge needed to build and apply statistical models. In part, the pressure to transfer estimates is due to the need to quickly gain easily identifiable information concerning an estimate, which may limit the appeal of sophisticated statistical approaches and models. Statistical approaches confer advantages such as gaining a more precise transfer estimate or estimate of transfer uncertainty. If precision is not necessary for the transfer and a more coarse description of uncertainty is sufficient, transferability assessment may be well served by previous qualitative approaches. Middle-ground may be found by applying simpler approaches (e.g., simple linear regression) versus more complex options (Ordination techniques such as non-metric multidimensional scaling (MDS) or bayesian modelling approaches). A merit of this framework is flexibility in how transfer errors are assessed, so that users may determine whether the effort required to predict a new estimate specific to the policy site is worth the gains in precision or description of

uncertainty. The case study will provide examples of different middle-ground approaches in order to further support this discussion.

3.4 Step Four: Total Transfer Error and Uncertainty Analysis

The final step of transferability assessment requires the synthesis of uncertainties and conclusions from previous steps to inform a transfer decision. Starting from step one, the user has recorded necessary assumptions which are now addressed. Some questions that may guide the uncertainty assessment may include:

- How strictly was the reference frame defined?
- How much uncertainty was allowed or introduced by methodological approaches or surrounded individual estimates?
- How representative were investigated intrinsic variables of core variables for the process? (i.e., were proxies or covariates used, or were some but not all core variables explored?)
- How are selected extrinsic variables representative of differences between sites?
- Were adjustments or transformations required to compare data?
- How much variability exists across the metadata? How does this compare to variability in terms of contextual characteristics similar to the policy site?
- [if conducted] How much variability in the data is explained by the model (e.g., R squared value)? What is the standard error around the predicted estimate?

After the user has considered all assumptions and sources of error (measurement, publication and generalization) the total error surrounding a transferred estimate may be judged. At this time, users may decide which estimates to transfer, or if another approach is needed for the intended assessment (i.e., to measure the estimate-of-interest at the policy site and forego estimate transfer).

As previously discussed in step two, a lower bound on total transfer error may be the error associated with the original estimate, or the measurement error. Sources of measurement error should be considered in addition to contextual similarity, which represents generalization error. For example, if a single study site is identified to have the highest context convergence with the policy site, and contextual assessment has identified that this will likely be more representative of the policy site than other estimates, users next must evaluate sources of measurement error. If at the original site this estimate represents an average, or was modelled it may have an associated variability metric (as evaluated in step two, e.g., as a standard error) which may directly be applied as variance to the estimate. The treatment of measurement error ultimately depends on the number and quality of estimates used in the transfer. If only one estimate is selected, determining measurement error may be more straightforward as in the previous example. However, when aggregating estimates across studies either as an average or if a transfer model was built, variability is introduced within and between estimates due to both measurement and generalization error. Therefore, further uncertainty analysis may be required to evaluate the total transfer error depending on if and how within-study variability was treated in the quantitative transferability assessment (i.e., propagation of uncertainty, see (Hoffman and Hammonds, 1994)). For example, within-study variance may have been included via weighting

based on variance in the transfer model if a fixed-effects or random-effects meta-analysis model was utilized (see Gurevitch and Hedges 1999; Lortie et al. 2013 for more information).

Publication bias has been previously discussed as another source of error which contributes to the total error surrounding transferred estimates. Publication bias represents the misrepresentation of study results and distribution of study site estimates due to the underreporting or delayed publication of non-significant results in the available literature (Rosenberger and Stanley 2006). It is difficult to assess the magnitude of publication bias however some methods exist such as funnel plots (Copas and Shi 2000) which relate significance or goodness of fit metrics to sample size or other precision metrics. Funnel plots permit visual inspection of the distribution of results with expected statistical distributions and are relatively easy to construct, however more advanced tests exist such as the funnel-asymmetry test (see Stanley 2008).

Assessing context similarity as an indicator or generalization error is a major focus of the framework. Therefore, conclusions whether qualitative or quantitative must be integrated with other sources of error to understand the total error of the transferred estimate. As previously mentioned, quantitative approaches may help predict a policy-site estimate that may capture some or all of the total error and therefore have representative confidence intervals and other statistical metrics. Qualitative approaches however, may leave the interpretation of multiple sources of error to the user. All assessment approaches, however, leave defining acceptable limits of error to the user. These may be related to the intended use of the transfer estimates, whether in litigation, policy evaluations, site assessments, or potential (i.e., predicted) magnitudes of effects on importance.

After total error is accounted for, users may synthesize assumptions and sources of error further through uncertainty analysis. The goal of uncertainty analysis is to identify the “errors, inexactness, unreliability and imperfection” (Wu, Jones and Li 2006, p. 44) inherent to ecological investigation and modeling. While many tests and approaches for uncertainty analysis exist depending on the approach used, general approaches which apply to many analyses concern error propagation, sensitivity analysis and validation (See Wu, Jones and Li 2006, chapter 3 for review).

Error propagation refers to how errors in input data or surrounding parameters affect model outputs. Output uncertainty from error propagation is assessed by determining how variances in model variables change the modelled output, and the variability in modelled outputs is a measure of the output uncertainty. Examples of methods used to assess error propagation include Monte Carlo simulation, generalized likelihood uncertainty estimation and sequential partitioning (see Wu, Jones and Li 2006 for descriptions). Similar methods may be applied in sensitivity analysis, a related term used to describe the investigation of how uncertainty in model outputs is related to sources of input uncertainty (Copas and Shi 2000; Yang 2011).

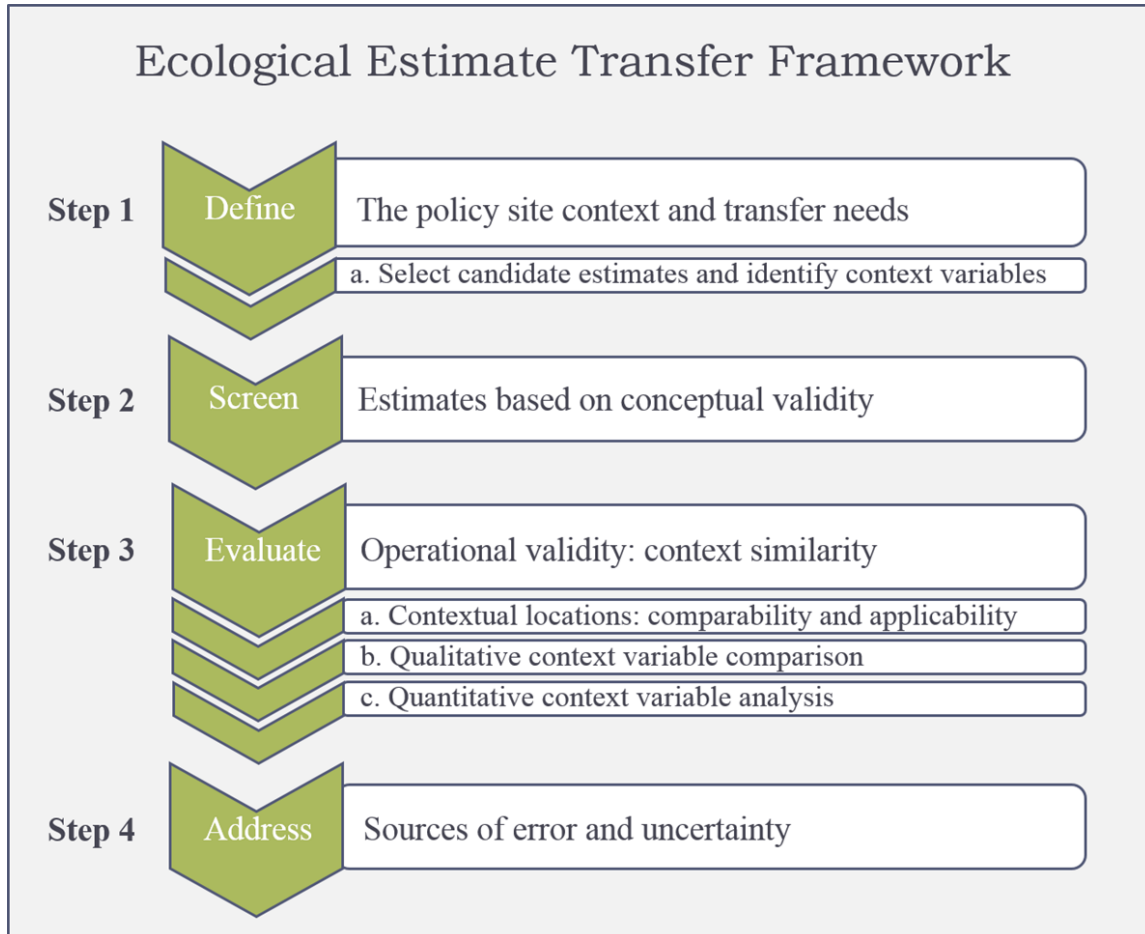
Sensitivity analysis quantifies the rate of change in a model when one of more input variables and parameters are varied while others are held constant (Wu, Jones and Li 2006; Boyle et al. 2009). Validation techniques, for example through bootstrapping or cross-validation may help elucidate the ability for a transfer model to predict outside of the metadata, or for selected samples in the metadata that were intentionally left out for validation (Bennett et al. 2013; Jakeman, Letcher, and Norton 2006; Rykiel 1996).

The treatment of cumulative errors is an important challenge for transferability assessment. In particular, how to reconcile trade-offs associated with selecting few or individual estimates that may have high contextual convergence but high variance or statistical uncertainties, or vice-versa (Johnston and Rosenberger 2010). It is ultimately important to know how the uncertainty associated with the estimate affects uncertainty in the overall ecosystem service assessment or model. This may be complicated for many ecosystem service models, for example linked economic and ecological models (Wainger and Mazzotta 2011).

Table 1: Benefit Transfer and Ecological Estimate Transfer Framework comparison. Steps from EPA (2000) guidance for preparing economic analyses (left) and the proposed framework for performing ecological estimate transfers (right).

Example Benefit Transfer Framework	Ecological Estimate Transfer Framework
1. Describe the policy case	1. Define transfer needs
2. Identify existing, relevant studies	a. Theoretical and empirical guidance
3. Review available studies for quality and applicability	2. Evaluate conceptual validity
a. The basic commodities must be essentially the equivalent	3. Evaluate operational validity & context comparisons
b. The baseline and extent of change should be similar	a. Evaluate comparability and applicability
c. The affected populations should be similar	b. Qualitative context comparisons
4. Transfer the benefit estimates	c. Quantitative context assessment
5. Address uncertainty	4. Assess total error and uncertainty

Figure 1: Conceptual diagram of the framework for ecological estimate transfers



4. A Structure for Ecological Contexts

The framework for conducting ecological estimate transfers relies on describing contexts in several ways. Latter framework steps (three and four) rely on defining contextual variables as analogously defined for benefit transfer. However, initial steps (one and two) rely primarily on defining contextual ‘reference frames’. The contextual reference frame is proposed to help users

identify appropriate and theoretically consistent contextual variables. Ecological processes may be studied across many different levels with different implications for their nature and existence. In this section, a more thorough definition and discussion of this structure is provided.

4.1 Theoretical Foundations

The concept of a contextual ‘reference frame’ emerges when defining the components of context that are broadly characteristic of different ecological systems, within which ecological processes operate. For example, the movement, abundance or habitat association of fishes on a daily or annual time scale are described using different variables as representative of different processes. In an estuary, fish distribution and abundance throughout a day might be more strongly related to salinity gradients and tidal flux, or food availability (Hartill et al. 2003). By contrast, on a monthly or annual scale, abundance may be explained more by species and life stage, where diadromous fishes move in and out of the estuary at various life history stages (Beck et al. 2001). Such differences fundamentally relate to the relationship between pattern and scale, specifically that observed patterns are dependent on the scale of observation, and therefore so are the mechanisms used to describe the pattern (Levin 1992).

The contextual reference frame is bounded by the levels of each of three categories of scale that broadly differentiate ecological processes (space, time and ecological organization) and is additionally described by the locational setting. The categories of scale represent the levels over which homogeneity either exists or is assumed, which serves as the basis for generalizability (Wu, Jones and Li 2006). Each category is intended to represent a distinct component of the reference frame. Scales, in space and time serve as the major theoretical

foundation for this structure. The effect of scale on the existence, nature and pattern of different ecological processes represent key issues in ecology (Pielou, 1977; Levin, 1992; Schneider, 2001). The consistent evocation of the concept ‘scale matters’ across biological and physical sciences (Schneider 2001) supports the approach taken by the transferability framework to define scale in order to assess scale compatibility as an indicator of broad contextual and ecological system similarity.

Scale refers to the extent relative to the grain of a variable indexed by time or space (Wiens 1989), where the grain is the minimum resolvable area or time period within some range of measurement (extent). Each category of scale represents a hierarchy of characteristic levels that have distinct extents and grain sizes. For example, in terms of ecological organization this is described as the movement from cell to individual, to populations, to communities and beyond (Odum 1959; Allen and Starr 1982). “Hierarchy theory indicates that, in general, the strength and frequency of interactions between levels decrease with distance.” (Wu 1999, p.377)

In addition to the effect of hierarchical levels of scale on the relationship between pattern and process, the effect of geographic and temporal proximity on the similarity of ecological observations is also well founded, of which spatial and temporal autocorrelation is of central concern to the field of spatial ecology (Gustafson 1998; Legendre 1993). The reference frame reflects the potential importance of both proximity and similarity across scales as transcendent principles that most broadly differentiate ecological systems and processes. Therefore, the components of contextual reference frames may serve as the basis for defining an ‘ecological meta-function’ for describing the levels of context that apply across ecological estimates.

It is proposed that by first recognizing the contextual reference frame, a basis to compare contexts at the system-level emerges. The ability to compare contexts can promote transparent and robust accounting as well as analysis of ecological estimate transferability.

“Patterns and dynamics at a given scale may develop from interacting lower-level units may also be imposed by large-scale constraints (Levin 1992). As a consequence, results from ecological investigations and predictions are critically affected by the scales addressed and their corresponding processes (Turner & Gardner 1991). In spite of this, the choice of the considered scales and aggregation levels is only rarely discussed explicitly in ecological investigations. This common neglect of scale-related questions may be the result of the scarcity of applicable methods for choosing appropriate levels of aggregation and for linking level-specific processes across scales (Kolasa 2005; Urban 2005).”(Meyer et al. 2010, p. 561)

Scaling issues, as resulting from the extrapolation or interpolation of processes across the hierarchical levels of scale have been extensively addressed in the ecological literature (see Wu, Jones, and Li 2006). The framework draws attention to the fundamental consequences that scale transformations may have on ecological processes by classifying scaling transfers as transfers which change the bounds of the reference frame. Transfers to different scale levels can be thought of as changing the size or scope of the reference frame, or ‘scaling’ by extrapolation or interpolation. For example, utilizing a measurement from one month to represent a year, or a stand of trees to represent an entire forest represent issues of scaling. By contrast, transfers between sites that are at the same scale levels have comparable reference frames, such a measurement from one stand of trees in California and the same measurement from a stand of trees in Oregon. Because of the fundamental consequences of transfers across different reference

frames, the framework identifies these transfers as a potentially greater source of generalization error and uncertainty because of the implications for the composition of intrinsic variables for ecological processes. Both the existence and the interaction of various processes may change depending on the scale of observation, which will be discussed in more detail in this section. However, while transfers between sites at comparable reference frames may help reduce potential generalization errors, this requirement does not eliminate generalization error. Differences between sites in terms of their location, or name of the scale levels, as well as contextual variable differences are additional sources of generalization error. Additionally, as discussed previously, *total* transfer error will also be affected by both measurement and publication bias.

The ability to generalize across scales and locations represents an important issue in ecology and to the transferability of ecosystem services estimates. The presented framework draws attention to this issue specifically by differentiating and highlighting contextual reference frame changes as major potential source of generalization error. Ecological processes may be more similar across comparable reference frames: “Parameters and processes important at one scale are frequently not important or predictive at another scale”(Turner et al. 1989). The structure for ecological context used by the framework relies on the premise that few ecological processes are scale invariant (Wu 1999) and scale effects may be profound (Wiens 1989). Because of this, the framework highlights consistency between reference frames of the study and policy sites prior to selection of context variables, to ensure that those processes generating the estimate-of-interest are also comparable across sites.

4.2 Contextual Reference Frames and Ecosystem Services

The following discussion represents a synthesis of prior thought and research relevant to the transferability of ecosystem service estimates across contextual scales to overview the scope of issues and practices. This serves to relate the status of current practices and issues to the proposed definition of ecological contexts, including contextual reference frame scale categories, levels and locations, to illustrate how holistic consideration of contextual reference frames and respective variables may increase the robustness of ecological transfers.

4.2.1 Spatial Scale

Spatial scale represents the physical dimension of the contextual reference frame, represented in both extent and grain, by metrics of length, height (or depth), area or volume. The hierarchy of spatial scale levels includes local and regional scales up to global (see table 2). In terms of physical scale, spatial transfers may therefore occur between sites that correspond in terms of spatial extent and grain, or at the same site at a different extent or grain, or both.

Transfer issues related to spatial scale are well represented inside and outside of the ecosystem services literature. Issues resulting from transfers across spatial scale levels in ecosystem services, via extrapolation, have been addressed most commonly by the ecosystem service literature (Helfenstein and Kienast 2014). This discussion has been prompted in part by the common practice to take literature values derived from one or more studies and apply it to a larger spatial extent in order to get an estimation of landscape or global total ecosystem service value (e.g., Li et al. 2010). However, other issues with transfers (e.g., at the same extent but with a different grain) may exist. For example, using an average value across a landscape or region to

represent units within the region (regionalization error), or using average representative units to represent other similar units within the same extent (sampling error).

Several authors have noted the issues associated with extrapolating from small-extent estimates to large extents (Eigenbrod et al. 2010a). For example:

“... original studies valued small changes in specific and localized components of individual ecosystems ... it is incorrect to extrapolate the value estimates obtained in any of these studies to a much larger scale, let alone to suppose that the extrapolated estimates could then be added together.” (Barbier et al. 2011, p. 173)

Yet, despite this, extrapolation remains a popular approach to generating estimates of ES production and value. This is because despite its limitations, it provides insight and value to things that are otherwise given a magnitude-less value or priority in decision-making (Costanza et al. 2014). Such studies may utilize what is sometimes called the proxy method, where an estimate derived for a particular land cover or ecosystem class is applied to all other units of the same class (Nemec and Raudsepp-Hearne 2013). Relatively little is known about how the errors associated with proxy-based methods might affect the inferences drawn from analyses because quantifying the impacts of such errors is difficult without comparisons using primary data. However, Some shortcomings of proxy-based maps for ES economic values have been addressed through sensitivity analyses (e.g. Nelson et al., 2009; van Beukering, Cesar, & Janssen, 2003).

Important guidance for the treatment of spatial scale related errors in ecological data in ecosystem service mapping come from Eigenbrod et al. (2010a, b). Eigenbrod et al. 2010 (a, b) discuss and quantify errors associated with the lack of accounting for spatial variability of

biophysical variables in proxy-based maps of ecosystem services. They compared field data for four terrestrial ecosystem services (recreation, carbon storage, biodiversity and agricultural production) and assigned production estimates to 14 land cover classes. They tested the relative importance of different components of generalization error by altering both the number and locations of grid cells used to obtain values for each land cover type. Errors investigated included: *uniformity error*, obtained by averaging all primary data values for a given ES that corresponded to a given land cover type; *sampling error*, obtained by taking a random sample of primary values and using an average to represent the value and; *regionalization error*, by averaging values based on 100 square kilometer grids in order to simulate variation by region. Poor fits were found between all maps and primary values and in particular for identifying hotspots or edges. However, the authors note that the difference between the proxy-based maps and the primary data only becomes important at more specific levels of inquiry:

“For example, if the goal of our study had been to identify whether the southeast of England had more biodiversity than the northwest, then we would have obtained the same answer using our proxies as using our primary data surfaces for biodiversity. However, proxies were completely unsuitable for selecting the top 10% of land area in England for biodiversity or for recreation.” (Eigenbrod et al. 2010a, p.383)

The results of the study by Eigenbrod et al. (2010a) help elucidate the risks associated with not only proxy-based mapping, but also sources of bias and error in how estimates are aggregated and averaged as inputs in other types of ecosystem service assessments. Scale mismatches in transferred values may also appear in ecosystem service models or in other calculation steps in ecosystem service assessments. For example, Grabowski et al. (2012) applied an estimate of the value of oyster reef for fish production from a biogeographic region (the

southeastern United States) and use it as an estimate for oyster reefs generally. Such evidence supports the framework's approach to distinguish estimates based on reference frames and to associate high transfer risk with transfers that require generalizations beyond the spatial scale of an estimates' reference frame in order to represent the policy site.

4.2.2 Geographic Proximity

In addition to transfers that extrapolate or interpolate information to different levels of spatial scale, transfers at the same spatial scale level may also occur. Therefore, the reference frame also relates to the geographic location, as a centroid independent of scale as well as the name that describes the location, which may refer to the scale. This component reflects transfer assumptions based on geographic co-occurrence or proximity and ecological theory which relates the similarity of observations due to their spatial relationships.

The first law of geography states that “Everything is related to everything else but near things are more related than distant things” (Tobler 1970). In his aim to build a simple but useful population model for Detroit, Tobler applied this logic to illustrate that while the population change between 1930 and 1940 in Detroit was a function of the city's initial population size, ultimately the population growth was also affected by the world's population, across all locations. Tobler offered that one could either build a model with all other cities' populations (e.g., 16 thousand variables), or use the global population as “a single surrogate”. Alternatively, Tobler proposed the law of geography and built a model to describe the growth of the population of one cell as a function of the population of neighboring cells. Spatial statistics and inquiries of spatial autocorrelation invoke the first law of geography commonly (Tobler 2004). Spatial autocorrelation refers to the dependency of observations in space (Legendre 1993). Spatial

autocorrelation is also commonly investigated as a statistical concern since it violates assumptions of independence. Tobler's law is also visible in studies of distance decay of ecological patterns and processes. Distance decay has been investigated within multiple scales such as local and regional spatial scales (e.g., Nekola and White 1999), as well as across organizational scales (e.g., species, population and community) (Morlon et al. 2008). Because of the way patterns of ecological processes may vary across geographic distance, it is regarded here as an emergent characteristic of contextual reference frames.

Explicit investigation into the relationship between geographic location and the values of ecological ecosystem services was not found during literature review. Researchers may commonly use geographic similarity to represent similarity across other study features but validations of this assumption were not specifically found. Extrapolations of scale may or may not involve a transfer in geographic location. In addition, transfers using the 'proxy method' as described by Eigenbrod et al. (2010b) where an estimate is applied to other units of the same land cover, ecosystem or habitat class may also involve a transfer across geographic location.

In the absence of site-specific information, investigators may assume that geographic proximity confers similarity without citing evidence for the comparability of estimates across space, or in comparison to change across other components of the reference frame. For example, Feagin et al. (2010) state that they prioritized estimates for transfer if they were close to the Galveston Bay area in their study of the changes in ecosystem services in salt marsh in sea level rise scenarios. However, despite the stated intention to describe changes in fisheries supply service due to sea level rise scenarios their chosen estimate is an average replacement cost derived from a study of restoration benefits, which may be less theoretically robust from both an ecological and economic standpoint. Ecologically, by using this approach equivalence between

habitat use/viability of naturally occurring units of marsh and constructed units of habitat is assumed. Economically, this approach may be less desirable to determine value than other methods since it is both the average cost of restoration projects instead of lowest cost and is a proxy of value (Woodward and Wui 2001; Brander, Brouwer, and Wagtendonk 2013). There was no comparison of this value with other candidates that were derived either using other elicitation methods, in comparable habitats, or for the desired service in general so it is unknown if these were explored alternatively.

Occasionally, estimates across geographic locations are presented alongside the chosen estimate. If direct comparison or justification was not provided, the reporting of multiple estimates from various extents or other contextual levels allows for geographic proximity-based assumptions to be generally evaluated. For example, a study of habitat-fisheries linkages in South Australia, McArthur and Boland (2006) provide a range of seagrass primary productivity estimates from studies conducted in Australia (120 to $700 \text{ g C m}^{-2}\text{y}^{-1}$) and compare them to reported global estimates of seagrass primary production estimates from coastal reef systems and non-tropical shelves ($890 \text{ g C m}^{-2}\text{y}^{-1}$ and $310 \text{ g C m}^{-2}\text{y}^{-1}$, respectively). The authors use the average value from Australian studies in their calculations, stating that the difference between the low and high value of the other systems is attributable to the inclusion of other sources of production such as phytoplankton and epiphytes in the estimate. While subtle, this comparison serves two purposes. First, it supports the Australian estimate as accurate or consistent across the literature. Second, it implies that using an estimate based on a different reference frame component, ecological organization (i.e., ecosystem type) would not change the assessment drastically.

4.2.3 Temporal Scale and Proximity

The ecological theory which supports spatial scale and proximity as reference frame components is nearly identical to that which supports the importance of temporal scale and proximity. Scales of spatial and temporal scale are often described as linked or coupled, meaning increases across one will likely lead to increases across the other (Levin 1992; Wu 1999). This is due to the scales over which processes become visible as well as the scales that we can measure spatial and temporal processes. The latter speaks to the general difficulties of fine-scale measurements over long periods of time or fine resolution temporal data over large spatial scales. However, some characteristics of ecological systems only emerge on larger scales. For example, patterns of global ocean thermohaline circulation or the formation of and movement of deepwater currents and upwelling systematically across the globe are observed on average over the course of decades to centuries, but on a yearly or finer temporal scale the movement of water at any given location around the globe is a function of wind, storms, eddies and other processes (Garrett 2003; Lozier 2010).

As with spatial scales, it is regarded as invalid to extrapolate beyond an investigated temporal scale without accounting for the effect of time on ecological processes (Legendre 1993). This is reflected in research design theory on representative sampling and valid statistical conclusions (Ramsey and Schafer 2012). For example, a study of seagrass carbon sequestration in an Oregon estuary taken weekly over a two month period from September to November 2013 would likely not be representative of annual carbon sequestration rates at the same site for the same year. Not only is this beyond the extent of the investigation, but seasonal and inter-annual variability for carbon sequestration is both theoretically founded (Mcleod et al. 2011) and empirically validated (Macreadie et al. 2013).

Temporal effects are less frequently cited or investigated as possible drivers of error with respect to transferred estimates in either the benefit transfer or the broader ecosystem services literature. However, issues of scaling, including extrapolations and interpolations as well as the effect of geographic proximity on estimate similarity may have many similar consequences with respect to time.

Wu, Jones and Li (2006) describe the concepts of extent, coverage and spacing with respect to both spatial and temporal scales each of which reflects the opportunity for regionalization, uniformity and sampling errors, respectively. Therefore, it becomes important to ask if firstly, candidate observations are at the same scope (extent and grain) as the policy site and secondly, if observations are representative of the entire temporal extent as well as opportunities for bias from sampling and uniformity error.

The proximity of estimates in time also has implications for their similarity either due to functional relationships and dependencies, or as a covariate. Observations may be autocorrelated in time due to processes which act at multiple temporal scales (Legendre 1993; Li 2005). Daily trends, such as tidal exchange may cause correlation in some observations, such as estuarine salinity on an hourly basis, since the salinity at a given time will be dependent on (or predictable from) the salinity one time step before or after the present (Ellis and Schneider 2008). Contextual variables that are either intrinsic or extrinsic to a given study, such as air or sea surface temperature, can be autocorrelated on daily, seasonal and annual basis. Processes also show autocorrelation at larger time scales, such as climate variability in the Pacific Ocean, which has roughly decadal oscillations between regimes, or global climate change over hundreds or thousands of years (Mantua et al. 2002). Therefore, proximity in time between a study-site estimate and the temporal setting of the policy or application site therefore may increase the

likelihood of similarity via autocorrelation for multiple variables. In addition, proximity in time may also confer similarities that are indirectly related. For example, most technological and methodological improvements in science are assumed directional in time. In their review of primary production estimates, Duarte and Chiscano (1999) attribute differences between their review and a review ten years prior due to both an increase in data and methodological advances. It is important to acknowledge the potential for such temporal shifts, however, since transfers between multiple papers may obscure temporal differences; McArthur and Boland (2006) compare their chosen primary production estimate range to a global range in comparable habitats, however the cited source (Pauly and Christensen 1995) calculated reported estimates from another source (De Vooy 1979) which in turn was based on a synthesis of previous literature.

4.2.4 Ecological Organizational Level and Classification

A third component of scale refers to the structure of ecological systems and similarly has a reference point or “location” which corresponds to the name or classification given to different levels of ecological organizational scale.

Ecological organizational scale follows hierarchy theory in ecology (Allen and Starr, 1982). Hierarchical structuring in ecological systems followed insight from biology that organisms are made of cells (Weinberg 1975) and other ecological levels follow such that organisms compose a population, groups of populations form a community et cetera up to levels of ecosystem and global levels of complexity (O’Neill, Johnson, and King 1989).

“Hierarchy theory suggests that ecological systems are nearly completely decomposable (or nearly decomposable) systems because of their loose vertical and horizontal coupling in structure and function.” (Wu 1999, p. 367)

Interpolations or extrapolations across different reference frames with respect to ecological organization occur when estimates either observed or modelled for one level (table 2) are applied to finer or coarser level. Transfers at the same reference frame level may occur when an estimate at a given level is used to represent other units of the same level, therefore at a different location, time or ecological classification. While most ecological levels in a hierarchy of ecological organization are clear and well-defined and correspond with ecological inquiries and disciplines (e.g., studies of populations versus ecosystems) others such as those which focus on guilds or food-webs may not be as clear. In addition, studies of many biophysical and chemical processes may only implicitly refer to levels of ecological organization. For example, primary production rates in seagrass beds may appear to refer to the species level, however rates derived may include production due to epiphytes or microalgae on the seafloor, which changes the level to that of the community (Cambridge and Hocking 1997; Jernakoff and Nielsen 1998). Furthermore, for transfers at the same ecological level, such as an ecosystem, differences within may be difficult to compare because of the large diversity of classification and naming conventions for these systems.

Ecological classifications, such as ecosystem, biome, or Land Cover Land Class (LULC) are sometimes used as the primary basis for the transfer for ecological estimates because they are perceived to represent consistency or homogeneity across ecosystem structure, function or both. For example, researchers may prioritize studies for transfer if they were conducted in the same or similar ecosystem type (e.g., McArthur and Boland 2006). This may be intuitive for very

divergent ecosystem types, such as wetland versus grassland, where there are fundamental differences in ecological setting (e.g., terrestrial vs. aquatic) but becomes less clear depending on the process or ecosystem types compared, such as specific classes of wetland or estuary (e.g., bar built or salt-wedge estuaries). In addition, the representativeness of the ecological classification may be confounded if there are other inconsistencies across dimensions of context or other variables. Furthermore, there are many available habitat, landscape and ecosystem classifications with varying connections to underlying processes, so the utility of a given classification to be effective for describing ecosystem service production may depend on the service and the scope of inquiry.

Extant classifications of ecological systems may require further validation and testing to test their utility for determining transferable relationships. Biome and ecosystem type classifications date back to the 1930s and 1940s and are based on climatic data, because this was the only globally available variable for extrapolation (Ramakrishna and Running 1996). Land use and land cover classifications have since been modified since their origin (Homer et al. 2007 e.g., the National Land Cover Database, NLCD) and are designed to differentiate terrestrial space based on remotely sensed geographic and vegetation characteristics (e.g., elevation, canopy cover). Therefore, the utility of popular and extant classification systems for transferability assessment depends on the correspondence between the classification basis and contextual variables that are meaningful for the process and estimate in question.

Recent investigations to generate a robust and consistent classification scheme of ecological habitats, similar to a chemical periodic table of elements, observe that while the relationship between structure and function of habitats for ecological functions is an empirical question, it is often unquestioned in applications predictive in nature (Ferraro 2013). In other

words, intuition about the structure and function of habitats is a good starting place for investigation, but predictive relationships of community usage and other ecological patterns require validation. Ferraro (2013) notes:

“When a community is quantitatively sampled in a statistically rigorous, unbiased, representative manner in an ecologically relevant space-time frame, tests for quantitative, periodic habitat-community patterns can be made under the a priori assumption that the community is defined appropriately. When, under these conditions, quantitative, periodic habitat-community patterns are found, the operationally defined habitat types are appropriate for the purpose in the spatial and temporal domain they were tested” (Ferraro 2013, p. 1543).

Investigations into these relationships are not uncommon in the ecological literature (e.g., Rabeni, Doisy, and Galat 2002), however, validation and investigation in contexts such as for ecosystem service supply or production are rare, if any exist. Others have noted the lack of ecosystem service-specific classification systems and have created ecosystem service driven classifications to fill this need (e.g., Townsend et al. 2014; Yapp, Walker, and Thackway 2010) however no single classification is predominant or has been broadly adopted.

There is strong ecological theoretical support for the proposed structure of ecological contexts as reference frames based on scale level and location across space, time and ecological organization. It is clear from both this theory and previous work in ecosystem services that both generalizations and transfers across dimensions may have significant consequences for transfer accuracy and reliability. However, previous discussion in the ecosystem services literature with respect to transfers across these different components of scale is not comprehensive. Where some

limitations due to differences in reference frames are well represented, including extrapolations of spatial scale and the use of LULC and ecosystem type as proxy, other issues such as transfers of information at the same spatial or ecological level are not, such as different geographic locations and ecological classifications. Extrapolations across time have received less attention as well. Perhaps most importantly, discussion concerning transfers across all or comparing consequences between these types of transfers is lacking.

Table 2: Contextual reference frame categories of scale and example levels. Hierarchical levels are represented from broadest (top) to finest (bottom).

Contextual Reference Frame Category	Example Reference Frame Levels
Spatial Scale	Global Regional Landscape Local
Temporal Scale	Geological Century Decadal Annual Seasonal Monthly Daily Hourly Instantaneous
Ecological Organization	Biosphere Biome Ecosystem Populations Individual Cellular

Table 3: Example contextual classification systems. Reproduced with author's permission from DeWitt et al. (2014).

Climate, weather NCAR Community Climate System Model Climate.gov PRISM	Fish & wildlife USFW Threatened and Endangered Species Critical Habitat NOAA marine critical habitat NOAA essential fish habitat mapper National GAP Species Data Portal Wildfinder Database BISON
Geology, soils USGS National Geologic Map Database USDA Natural Resources Conservation Service	Ecological regions Conservation GIS Data including Ecoregions Ecoregions of North America ESRI Global Ecological Land Units Marine Ecoregions of the World (MEOW) Freshwater Ecoregions of the World (FEOW) The Ecological Site Information System (ESIS)
Elevation, topography National Elevation Dataset World Ocean Atlas	Human infrastructure TIGER National Transportation Atlas Databases 2014 NLCD 2011 Percent Developed Impervious Homeland Infrastructure Foundation Level Data (HIFLD) Facility Registry Service
Hydrology NHD, NHDPlus HydroSHEDS Hydrologic Unit Maps	Environmental data atlases USEPA EnviroAtlas ESRI Living Atlas
Land & seafloor cover MRLC National Land Cover Database (NLCD) National GAP Land Cover Data Portal Coastal & Marine Ecological Classification (CMECS) National Wetland Inventory	
Vegetation Length of growing period USDA Plant Hardiness GIS data LANDFIRE	

5. Case Study: Transferring Estimates of Benthic Microalgal Primary Production

In the GecoServ valuation database review, a few common estimate transfers were identified within studies of habitat-fisheries linkages. Habitat or ecosystem type was used as a proxy to apply ecological estimates to maps in several studies (five of twelve instances), however parameter transfers were found to be slightly more commonly used, as derived from the literature (in seven of twelve instances). Studies which utilized parameter transfers represented un-calibrated estimates (i.e., were not adjusted for policy site conditions). Furthermore, two types of parameters were commonly transferred, either estimates of the natural mortality rate for fishery species or estimates of primary production rates of the habitat of ecosystem types studied.

Primary production estimates were utilized in studies where secondary production is calculated from or related to the extent of primary production (five instances). Approaches that required natural mortality rate estimates were less common (two estimates). Natural mortality rate estimates were used in individual-based population models or other predictive modelling approaches, similar to stock assessment models (Clark 1999). Because parameter transfers of primary production rates were the most prevalent, this was selected for further investigation in a case study to explore the utility of the framework approach for evaluating ecological estimate transfers.

A general form of this relationship relates the contribution of benthic microalgal primary production (BMPP) to secondary production, as represented by fisheries value (or alternatively in metric tonnes of catch). An example of a simple model is provided in equation 1 (McArthur and Boland 2006):

$$\text{Benthic microalgal primary production value } (\$ m^{-2} yr^{-1}) = \frac{\left(\frac{PP_{bmpp}}{PP_{total}}\right) * \text{Fishery Value}}{A_{bmpp}} \quad (\text{Eq.1})$$

$$PP_{bmpp} = PPR_{bmpp} * A_{bmpp} \quad (\text{Eq. 2})$$

$$PP_{nb} = PPR_{nb} * A_{nb} \quad (\text{Eq. 3})$$

$$PP_{total} = PP_{bmpp} + PP_{nb} \quad (\text{Eq. 4})$$

Equation one estimates the value of BMPP by relating the magnitude of BMPP (PP_{bmpp}) to total primary production within the site (non-benthos habitats, PP_{total}) (in terms of areal extent). To derive indicated parameters, the calculation requires primary production rate estimates for the specific habitat type (the benthos, PPR_{bmpp}) and the areal extent of this habitat type (A_{bmpp}) (Eq. 2). In addition the primary production rate of non-benthic habitats (PPR_{nb})

and extent (A_{nb}) is needed (Eq.3) to estimate total primary production for the site (PP_{total} , Eq. 4). The ratio of BMPP to total primary production is then multiplied by the total value of the fishery (Fishery value, in landings per year). The resultant annual value of BMPP is divided by the benthos area to derive the value per unit area. The case study focuses on the hypothetical transfer of the benthos-specific estimate of primary production rate (PPR_{bmpp}) as used in equation two.

5.1 Step One: Study Purpose and Data Selection

In this example, a hypothetical study seeks to quantify ecosystem services production by the Columbia River estuary, Oregon, USA (i.e., the policy site). For this assessment an estimate of BMPP is needed to calculate the contribution of primary production to secondary production. Benthic primary production is generated by microphytobenthic algae, which are the microscopic algae (primarily diatoms) and cyanobacteria that reside on the seafloor. Most of the microphytobenthos reside in the top few millimeters of shallow unvegetated habitats (e.g., mudflat, river channel bottom) but also reside in other submerged aquatic habitats such as salt marsh and seagrass beds (Macintyre, Geider, and Miller 1996).

First, the contextual reference frame is defined for each level of the three contextual scale categories. The spatial scale is defined by both the extent and grain of the policy site. The spatial extent of the policy site is specifically the lower 410 km² of the Lower Columbia River estuary (Lower Columbia River Estuary Partnership 2007). The spatial grain is the represented habitat type, in square meters, based on the resolution of the hypothetical fisheries model (Eq. 1). The temporal extent of the study is a year and the model is not temporally distributed, therefore the grain and extent are the same. In terms of the ecological scale, the extent is the habitat type

specified, or the benthos, as the homogenizing unit that differentiates this rate from other types of primary production rates, such as in the water column (phytoplankton) or from submerged aquatic vegetation (macrophytes). The ecological ‘grain’ is the community, since BMPP is due to the photosynthesis of multiple types of organisms.

The descriptions provided for each category of scale represent the reference frame levels and in addition can be described by name or “location”. The geographic location is equivalent to the extent name, the Columbia River estuary, Oregon but is also represented by its latitude and longitude, approximately 46.2° N and 123.8° W. The temporal location is 2015 and the ecological location may be described in several different ways. The name of the reference frame level, ‘community’ is the microphytobenthos. As part of the geographic name, the ecological setting is partially described as an estuary. A detailed classification of the terrestrial and aquatic ecosystem of the estuary exists, The Columbia River Estuary Ecosystem Classification, (CREEC, Simenstad et al. 2011). The ecosystem classification provided by Simenstad et al. (2011) provides an in depth description of the hydrological and geomorphic setting of the policy site, providing six hierarchical levels of classification across “ecosystem province, ecoregion, hydrogeomorphic reach, ecosystem complex, geomorphic catena and primary cover class” (p.11) The classification uses and adapts currently available classifications for its use, including EPA-adopted ecoregions (see Omernik 1995, 2004). While many of the higher levels of the classifications describe primarily the terrestrial setting surrounding the estuary, the Ecosystem Complexes and Catena classification components map several depth regimes (e.g., deep channel, permanently flooded, side channel). Deep channels were classified as areas corresponding to the fourth quartile of depths, and correspond to greater than 8 meters (or approximately 26 feet).

5.1.1 Theoretical and Empirical Guidance

After the policy site reference frame is determined the next step is to review the literature to identify study site estimates and contextual variables. A previous review of 106 published BMPP rates (Cahoon 1999) was identified that could provide a source of globally-distributed study site estimates for transferability assessment. The literature review in transferability assessment is intended to balance enlightened inquiry and full meta-analysis as dependent on user needs. This is in recognition that transfers are potentially more likely in either time or budget limited policy applications, data-intensive ecosystem service assessments, or other applications which must prioritize analytical efforts. It falls to the user to decide how much understanding of the given process is needed, for which identification of the contextual reference frame and the scope of the study will assist with. In this example, differences across contexts are assessed broadly, across global locations. This enables examination of broad trends across greater ranges of contextual variables and also to maximize the size of the available dataset. In addition, this assists in the assessment of the relative importance of general contextual variables, such as geographic proximity and ecological classification which may help to inform the basis of selecting a subset of estimates to transfer (i.e., whether to select estimates from geographically proximate locations or from more similar ecological classifications). Depending on the results of a broad scale inquiry, users may decide to investigate further, collecting more information and defining more detailed contextual variables or can make a coarse, but informed transfer decision based on the initial broad assessment.

In a broad assessment, such as across global locations, literature reviews or previous meta-analyses are a good place to identify core intrinsic contextual variables and relevant extrinsic variables. Therefore, the data source (Cahoon 1999), another review (Macintyre,

Geider, and Miller 1996) and a more recent update (Cahoon 2006) were primarily used to identify relevant intrinsic and contextual variables. Basic photosynthetic physiology yields insight that intrinsic variables may be defined as light intensity, biomass (chlorophyll a) and temperature (Cahoon 2006). Therefore, factors and processes that affect core intrinsic variables are also important to consider. Such covariates are described as light attenuation (by both overlying water and sediment) and the degree of patchiness in benthic microalgal abundance in both space and time (Cahoon 2006; Macintyre, Geider, and Miller 1996). It is also expected that BMPP rates correspond across latitudinal gradients with increased seasonality at high latitudes (Macintyre, Geider, and Miller 1996). Essential nutrients, such as nitrogen and phosphorous are not described as limiting factors for BMPP and are therefore not included as a core variable for the process. This is described by Cahoon (1999, p. 71):

“The nature of near-bottom water flows in neritic area, which are driven by wave action, lunar and wind tides, internal waves, and other currents, ensures a frequently turbulent regime that... prevents nutrient depletion in the near bottom zone.”

Therefore, light, temperature and biomass emerge as key core intrinsic variables where factors or elements that affect these factors represent other potential intrinsic or extrinsic contextual variables. Extrinsic variables might include the site depth, climatic regime, or latitude which may affect the quantity and intensity of light reaching the seafloor. Other possible variables which can locally affect BMPP rates include the specific microalgal species composition, predator densities and predation rates on the macroalgal community, storm intensity and frequency at the site (Cahoon 2006; Macintyre, Geider, and Miller 1996).

5.2 Step Two: Conceptual validity

Initially, all estimates included in the case study were assumed sufficiently valid for the transferability exercise. Each study was previously included in a published meta-analysis (Cahoon 1999) and represents a large pool (86 studies) so that differences in methodological technique and other features affecting conceptual validity may be explored for impact later in the transferability assessment. However, this step may be necessary to revisit when selecting final or individual candidates for transfer to assess sources and magnitudes of measurement error.

5.3 Step Three: Operational Validity and Context Assessment

5.3.1 Evaluate Comparability, Applicability and Representativeness:

In step three, the applicability, comparability and representativeness of the pool of study site estimates for the policy site context is addressed. First differences in study site estimate descriptions and units are reconciled to allow for subsequent comparison. Then, representativeness is evaluated by comparing contextual scale levels, locations and contextual variable ranges to the policy site to ensure adequate representation.

Estimates reported by Cahoon (1999) are in equivalent terms and units. Estimates are representative of gross primary production rates versus net primary production rates (GPP and NPP, respectively). However, it is noted that different methods yield estimates that are slightly in between true GPP and NPP, which refers to the difference between the entire photosynthetic production of organic compounds (GPP) and the amount that results in growth (NPP)

(Macintyre, Geider, and Miller 1996). All estimates are reported as annual rates per square meter of habitat ($\text{g C m}^{-2} \text{ yr}^{-1}$).

Estimates also have comparable contextual variables. Cahoon (1999) reported location, depth (or range) of measurement, methodology, publication information, and climatic regime information alongside each annual BMPP rate estimate (see appendix 1). Estimates that did not have all contextual variable information available (e.g., methodology or location was missing) were not included in further assessment. An assumption required for analysis was that depth ranges, when a single estimate was reported, could be represented adequately by the median reported depth.

Estimates are assumed to have been measured at comparable reference frames. Estimates are assumed to represent equivalent spatial scales since no large-scale averages, such as continental or global averages, exist in the data set. Estimates are generally identified by site-specific names (e.g., San Antonio Bay, TX) however more general, regional identifiers are also provided (e.g., Danish fjords). Specific extents and resolutions of study site observations are not reported. Further study-specific investigation is needed to determine the scope of individual studies as representative of studied ecosystems (i.e., the extent of each study respective to the environment type and scale identified). Ecologically, included studies are all at the community level, where total primary production is measured resulting from assemblages of microalgae of various species (mostly species of diatoms but sometimes cyanobacteria, chlorophytes among others, Cahoon 1999). Temporal reference frames were adjusted by Cahoon (1999) to be comparable. Many estimates were converted from hourly (g C hr^{-1}) or daily (g C day^{-1}) BMPP rates using static conversion factors. This adjustment required several conservative assumptions:

“Shaffer & Onuf (1985) discussed proper methods for deriving daily, monthly and annual estimates of benthic microalgal production, but few of the published studies [used in this meta-analysis] satisfy the criteria they use for better estimation methods. Consequently, I use several conservative assumptions (10-h production days, production years of 365, 270 and 90 days in tropical, temperate and polar zones, respectively, and a conservative formula to calculate annual production.” (Cahoon 1999, p.54)

There are limitations to the approach used by Cahoon (1999) to standardize the dataset for comparison, but are assumed to be sufficiently correct as reported for the present step in order to proceed. However, errors in these transformations could lead to biased and inaccurate estimates. Thus this assumption is carried forward into the later uncertainty analysis (step four).

Based on the reference frame descriptions and in acknowledgement of required assumptions, the estimates are assumed applicable and comparable to each other and to the policy site. Next, representativeness is addressed to evaluate the coverage of the study site “locations” and variable ranges relative to the policy site. This step constructs an “application niche” in order to gauge how proximate available estimates are to the policy site spatially, temporally and ecologically and also across other available variable ranges.

Two approaches were used to explore the application niche of BMPP estimate contextual variables. Geographic points were attributed to each location by estimating the centroid, using the site names as provided and Google Earth. Then, latitude and longitude data were used to create a map in ArcGIS (figure 2) to visualize the distribution of estimates across the globe. Second, a radar plot was constructed in Microsoft Excel to compare estimates based on contextual variables reported in the Cahoon (1999) study (figure 3). Selected contextual

variables were limited to those that were both reported, those that could be compared between estimates, or those that could be geo-referenced. These included the depth (or median depth) where the estimate was collected, measurement approach, the climatic regime (temperate, tropical or polar), date of study publication, and the location as represented by an estimated latitude and longitude. In addition, other classifications for marine and coastal environments were identified using a GIS referenced database (Marine Regions, Claus et al. 2014).

Two applicable ecological and biogeographic classifications for coastal and marine systems were added as potential contextual variables that are finer than climatic regime. Marine Ecoregions of the World (MEOW) is a four-tiered hierarchical classification system for coastal ecological systems (i.e., nearshore). The coarsest level of MEOW (“Latitudinal Zone”) corresponds nearly identically to the climatic regime classification provided alongside the BMPP estimates by Cahoon (1999) (only one conflict occurred near a boundary in South America). Therefore, this classification was not used in the analysis because it was redundant. The “Marine Realms” level of the MEOW classification consists of twelve realms worldwide, ten of which were represented in the dataset. A finer classification of World Seas, a biogeographic classification was also used and identified 24 classes in the dataset. The finest level of MEOW, consisting of ecoregions was not applied, because of the accuracy of the BMPP site locations was insufficient to confidently place those sites within specific ecoregions.

The map (figure 2) and radar plot (figure 3) illustrate the overall patchiness of the dataset. The map illustrates the concentration of data in mid-latitude temperate regions in the northern hemisphere, off the east coast of the United States and Western Europe, in particular, with representation on all major continents, but lowest representation in polar latitudes and in South America and Asia. In terms of geographic location, three estimates are from Washington State

(Grays Harbor, False Bay, Chapman Cove). Comparing the temporal location, the studies from Washington and most others in the data set are from the mid-1980s, up until mid-1990s. The temporal extent and grain of the data in Cahoon (1999) are 1959 to 1996 and annual, respectively.

Tropical and polar observations are limited in comparison to temperate locations. The radar plot illustrates that approximately two-thirds of the data were collected in the temperate climatic regime (as designated by Cahoon (1999)), with less than a third of observations in tropical regions and just a few studies in polar zones (figure 3). The radar plot adds an additional variable to the information derived from the map by allowing the depth of observation to also be compared. Depth ranges vary, across sites and climate regimes, however a greater proportion of measurements from tropical sites come from depths greater than 0 (representing intertidal measurements) whereas roughly half come from intertidal observations in temperate studies. Longitude and date of study were not compared on a radar plot. Longitude was not included since it lacked a theoretical foundation (i.e., not a proxy alone) and the numerical range that was not as easily comparable to the magnitude of depth, absolute latitude and depth. This is a limitation of radar plots, since value ranges that are very different between values will skew apparent variability. For example, absolute latitude ranges from 0 to 90, however longitude varies from -180 to +180. By increasing the scale of the radar plot to include the extreme ranges in longitude, differences in absolute latitude, while meaningful, appear disproportionately less variable. For general representation, such as in the current example, ranges and values that are similar will elucidate major trends well, but most accurate representation requires values and ranges to be standardized. For example, the climatic regime categorical variable was standardized to a range for comparison.

From the radar plot and map of the estimates it can be concluded that while distributions globally and across depths are patchy, context variables that are most closely aligned with intrinsic drivers, such as depth, absolute latitude and climatic regime (because of their effect on light availability and intensity) are representative of the policy site. The latitude of the Columbia River estuary, at approximately (46.2°) N is similar to Grays Harbour (46.9°) and Chapman Cove (47.22°), Washington. In addition, sites in Italy and France are also at comparable latitudes (45.6° and 45.8°, respectively). The radar plot shows that many sites at similar latitudes are represented. A range of depths in temperate latitudes is also represented from the intertidal to 35 meters depth (the total range of the dataset extends to 60 meters depth). The total depth of the policy site extends to 60 meters depth however median depth is approximately 30 meters and mudflat and unvegetated bottom habitats are mostly intertidal (CREEC, Simenstad et al. 2011). Therefore, while the specific geographic location is not represented, the coverage and correspondence of the available contextual variables to the policy site characteristics do not indicate that the site is beyond the extent of the application niche for BMPP estimates and adequate representation across contextual variables exists.

5.3.2 Qualitative Context Comparison

Next, both qualitative and quantitative approaches may be utilized to explore and analyze the estimates in terms of their contexts. Scatterplots and histograms of the data with respect to the numerical estimates permit insight into the relationship between estimates' numerical values and selected contextual variables. If quantitative assessments will be conducted, this step may assist with the evaluation of assumptions and conditions required for statistical testing. While there is significant variability, declining trends in both production by depth and across absolute latitude (from high to low latitude) are visible (figure 4). Single observations for some

measurement methods, such as a combination of oxygen flux and carbon isotope uptake ($^{14}\text{C}/\text{O}^2$) seem to have very different values. However, the most prevalent methods, carbon isotope flux (^{14}C) and oxygen flux (O^2) (94 percent of observations) average values and standard deviations do not seem to vary substantially. The combined oxygen flux (O^2) and ^{14}C radiolabelling technique appears to yield different average value, however as an observation from the Great Barrier Reef and as one of the few tropical sites may not be indicative of bias. Estimate values seem to be fairly equally distributed across publication dates, which indicates that possible changes in methodological accuracy or other bias over time are not observable in the date ranges, however as previously noted more recent estimates up to the present do not exist in the data set.

From these initial simple visual explorations one may observe that the climatic regime, latitude and depth variables appear to correlate with changes across the observed BMPP estimates.

5.3.3 Quantitative Context Assessment

In order to explore the combined effects of contextual variables and derive transfer estimates for the policy site two statistical methods were utilized, specifically analysis of variance (ANOVA) tests and a Classification and Regression Trees (CART) analysis. These methods were used to explore the importance of the variables as previously identified in a univariate approach (using the former), and also to demonstrate potential multivariate relationships (using the latter). In addition, these methods illustrate potential differences across models of different complexities.

Individual one-way ANOVA models were constructed in R for three contextual variables: absolute latitude, depth and climatic regime. ANOVA models were not constructed for

World Sea or MEOW Realm classifications because of the small number of replicates within many groups. In order to compare differences between groups, continuous variables such as absolute latitude were compared at intervals of 10 degrees. Depth intervals followed breaks specified by Cahoon (1999) which distinguishes intertidal sites (“0” depth) from subtidal sites. Subtidal sites were grouped from less than zero to five meters, from 5 to 20 meters and greater than 20 meters. Box and whisker plots of the data and results of each test (figure 4) conclude that the most significant difference across groups are observed across climatic regimes (P-value <.001), however significant differences across latitudinal classes were also observed (P-value = 0.0013). Some evidence of differences across means was found for depth classes (P-value =.0318). A nonparametric test (Kruskal-Wallis) was also used in order to investigate the effect of deviations from the ANOVA assumptions of normally distributed residuals and independence between groups. The results of these tests showed that conclusions of significance were robust across all models. Possible transfer estimates based on climatic regime (temperate) or absolute latitude (40° to 50°) of the policy site result in mean estimates of approximately 89 or 70 g C m⁻² yr⁻¹, respectively.

The CART model was constructed in JMP using all contextual variables explored individually in the ANOVA models. The CART model explains variation in a single response by recursively splitting the data into more homogenous groups, using a single continuous or categorical predictor variable at each branch. For continuous response variables, each group produced by the model is represented by a typical value of the response variable (the mean), the number of observations that define it and the specific values of the explanatory variables used (De’ath and Fabricius 2000; McCune and Grace 2002). The final CART model (lowest AIC) used absolute latitude and depth to define the number of splits and final categories (figure 6).

The CART model is read like a dichotomous key, where first the data is split into two groups based on absolute latitudes greater or less than 18.7° , and then splits the representative data based on either depth or latitude. In total, the model breaks out the data into eight different groups, which are defined by the final node and all of the nodes which precede it. The policy site is entirely predicted by its absolute latitude, where it falls within the group defined by absolute latitudes greater than 38.7° and less than 55.53° which predicts a mean value of $70.7 \text{ g C m}^{-2}\text{yr}^{-1}$ (figure 6). Conversely, tropical locations (absolute latitudes $<18.7^{\circ}$) are classified by absolute latitude and depth. After the first split, locations at latitudes less than 18.7° are split three more times based on depth breaks of less than or greater than 2.5, 7.5 and 25 meters. Each final node predicts declining productivity with depth.

Overall, the use of quantitative approaches lends further statistical insight into the individual (ANOVA) or combined (CART) explanatory power of the contextual variables. Use of qualitative approaches may persuade a user to transfer an average value based on similarity across a climatic zone, latitudinal range and depth gradient. Multiple one-way ANOVAs illustrate that climatic regime and absolute latitude groups serve as the first and second most statistically significant categorizations of the data. Finally, CART analysis uses a data-driven approach to define categories that explain the most variation in the dataset. For the policy site, the CART model creates a group nearly identical to the group specified in the absolute latitude model and therefore predict similar policy site estimates.

5.4 Step Four: Total Transfer Error and Uncertainty Analysis

Several assumptions were made in earlier steps that were noted to potentially affect the accuracy of the BMPP estimates or the context variables. These assumptions thus affect one's confidence in the transferred BMPP estimate. These assumptions included:

1. The reference frames of the estimates and the policy site were comparable.
 - Temporal scale transformation was accurate: Few estimates in the data set across studies did not require a transformation across temporal scale (n=12).
 - Comparable spatial scales are represented: the data did not report spatial extents or grain of individual estimates, depth ranges are represented by median depth.
2. The estimates were representative in terms of both location and variable ranges.
 - Temporal locations were representative of the policy site: all included studies are well over a decade old.
 - Changes in ecological community composition (ecological “location”) are insignificant: differences with respect to species compositions between sites was not directly represented.
 - Geographically proximate observations and at similar latitudes convey sufficient coverage to be representative of the policy site conditions.
3. Proxies are able to represent intrinsic contextual variables: light intensity and availability was represented by depth, latitude and climatic regime.
4. Measurement techniques may produce systematic differences.

5. Assumptions required for statistical analysis: intra-study variability, effect size comparisons and other sources of statistical uncertainty.

The first assumption addresses potential differences in estimate reference frames, relative to each other and the policy site. The temporal scales required transformations to be comparable and the spatial scales were assumed equivalent. Both assumptions introduce potential generalization errors, such as sampling and regionalization error (Eigenbrod 2010 a,b). The representation of spatial scale in the data was only by depth, other information on the geographical extent or grain of individual observations within studies was not reported. Therefore, it was assumed that these differences did not affect the underlying process and that the representation of depth ranges by median depth would be sufficient. However this assumption may have contributed to the ability to assess the correlative effect of depth on BMPP rates across locations. As shown by the ANOVA model for depth, the overall relationship between BMPP and depth was not as strong as one might have expected theoretically (Townsend et al. 2014). However, declines in productivity were predicted for low latitude observations in the CART model.

The second assumption refers to the ability for the data set to capture the policy site context, where it was assumed that sites from the same biogeographic region (Washington) and latitudinal range could sufficiently represent the policy site, the Columbia River estuary, spatially, temporally and ecologically. The scale 'location' least well represented by the data is perhaps time, since there is nearly a 20 year gap between the most recent estimates in the dataset and the policy site location. The stability across methods used and BMPP estimate ranges across publication dates over 40 years is an indicator that generalizability risk due to this lack of correspondence (figure 4) is low, however it is not conclusive.

The third assumption refers to the ability for the available contextual variables to represent core intrinsic variables and therefore be useful for relating differences in estimate values across sites. Because only proxies of intrinsic variables were available the ability to capture differences between estimates may be reduced.

In addition to uncertainty and error introduced through the previous assumptions, each individual study had measurement error associated with it. As previously addressed, differences in measurement technique were not detected in the assessment but could be masked by the variability of the dataset as a whole. Other error, as represented by variability around individual estimates was not reported and therefore is not addressed. A more comprehensive meta-analysis could collect individual study data in order to weight estimates relative to sample sizes and variability (see Pigott 2012).

The modelled results represent several important indicators for the total transfer error (i.e., including both measurement and generalization error). Information available for the models includes the strength of evidence for observed differences between groups (i.e., not due to chance alone, P-value), the fit of the data to the model (R^2) and the variability or error around group means (standard deviation and standard error). Potential transfer estimates from each model based on the policy site characteristics are in table 4. Conclusions based on differences in statistical evidence among groups and across models should be balanced with previously identified limitations of the dataset, since these are not captured directly within the data. However, up to this point there is little indication what the effect of previous assumptions and transformations may have be for the reliability of the data and models. Validation testing, or tests of convergent validity as described in benefit transfer, are useful for evaluating such assumptions and transfer errors comprehensively (Rosenberger and Phipps 2007).

Several estimates were removed from the Cahoon (1999) dataset prior to transferability assessment in order to assess the ability for different approaches to represent values outside of the training data. One validation site represented the actual policy site, the Columbia River estuary. Percent transfer errors (PTEs) were calculated in two different ways to explore the magnitude of the effect of the temporal transformation on the data and also the ability for the models to predict at the policy site.

Five estimates were selected from most and least represented parts of the data set. Two estimates from Oregon, Netarts Bay and one from the chosen policy site, the Columbia River estuary were included as “well represented” points, since many other sites from temperate, mid-latitudes and shallow depths were in the dataset. In addition, another temperate site from the Netherlands (Ems-Dollard) was also selected. Estimates from Manukau Harbour, New Zealand and McMurdo Sound, Antarctica were selected as unique or less well-represented sites. The New Zealand estimate represents the only site from New Zealand in the data set, whereas one other Antarctic site was represented by the remaining data.

Percent transfer errors (PTE) were calculated to compare the performance of the models for predicting the validation site estimates. To calculate PTE, the original study estimate is used as the approximation of the true policy site estimate. The difference between a transfer (study site) estimate and a policy site estimate is divided by the policy site estimate and multiplied by 100 to calculate PTE (Rosenberger and Phipps 2007).

The results of the validation test illustrate that generally all of the models performed better for well-represented sites than for the New Zealand or Antarctic sites (figure 7). Interestingly, the more complex CART model only performed best to predict the validation

estimate that represented the policy site. The ANOVA models performed best at each of the four other validation sites. The depth ANOVA model performed best at the Netherlands site, however on average it performed the worst for all sites (average PTE = 52.5). On average, the ANOVA climatic regime, absolute latitude and CART models performed similarly (average PTE = 41.7, 42.4 and 44.3, respectively). Overall, the validation sites help illustrate how transfer errors may vary across models and sites, depending on the representation in the dataset.

More extensive validation testing is required to discern the best or most reliable model, especially since performance was found to be fairly variable within and across models. In addition, since validation estimates come from the same transformed data set, estimates which were transformed in this way are not as representative of the ‘true’ site value as we untransformed values, which represents an additional source of generalization error. Therefore, the validation exercise cannot gauge the difference between these true values and the transformed representations of true estimates in the dataset. However, the validation site estimate at the Columbia River Estuary was not transformed from hourly or daily rates which allows for a better approximation of the total transfer error for the present case study.

To explore the potential effects of the temporal generalization on total transfer error, the original Columbia River Estuary study data (Mcintire and Amspoker 1986) was used to estimate a hypothetical transformed estimate for the study, henceforth referred to the transformed policy site estimate. The mean hourly BMPP estimate ($41.6 \text{ mg C m}^{-2} \text{ hr}^{-1}$) was converted to an average annual estimate by multiplying the estimate by ten hours, to represent the assumed number of hours of primary production in a day, and by 270 days to represent the number of days in a year that production is assumed occur in temperate latitudes (Cahoon 1999).

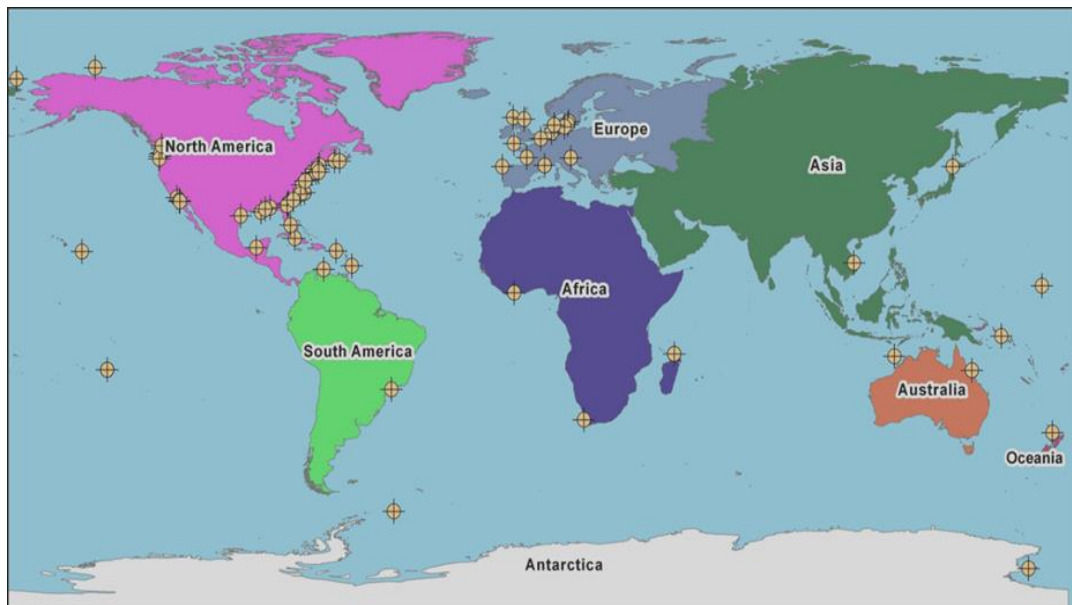
The transformed policy site estimate was calculated to be $112 \text{ g C m}^{-2} \text{ yr}^{-1}$. The ‘true’ value is represented by the original annual estimate reported by McIntire and Amspoker (1986), $72 \text{ g C m}^{-2} \text{ yr}^{-1}$. There is a 55.6 PTE between the original policy site estimate and the transformed estimate. This difference may represent a conservative estimate of the bias introduced by the transformation since the average hourly estimate represents data that spanned both a year and the entire extent of the lower Columbia River Estuary. This estimate of bias is useful for illustrating the generalization error at the policy site, however does not assess the effect of the transformation on the remainder of the dataset, for which replication is needed. PTEs were calculated using both policy site estimates, the transformed and the original study value, to represent the true policy site across model predictions (table 4). This serves to explore the potential effect of the transformation on total errors between models.

The uncertainty assessment helps illustrate that if the data from the original study of the Columbia River Estuary was transformed like much of the rest of the data set (30 of 106 estimates total) the ‘best’ and ‘worst’ models would change, as opposed to comparing the predicted value to a value that was not transformed. The best models for predicting the true value at the policy site are the CART and absolute ANOVA latitude models, since they split estimates into similar groups (PTE= 1.8, 2.8 respectively). A simple average of estimates from Washington ($n=3$) also results in close predictions. For the transformed policy site value, the CART model performs the worst (PTE= 36.9) and the best model is the ANOVA depth model (PTE=7). Overall, the models performed better for the true policy estimate than the transformed estimate (average PTE= 16.2, 26.5).

Finally, users may select a candidate transfer estimate and specify a range of values to determine the sensitivity of the final ecosystem services assessment or model results. Ranges of

values around the estimate may be represented by the standard error, or by specifying a broader range, to reflect other uncertainties. As discussed previously, depending on the type of model or assessment both simple and complex approaches are available for conducting such sensitivity analyses. Remaining data necessary for the habitat-fisheries linked model proposed in this case study was not collected and therefore this sensitivity was not explored. However, future assessments may explore this in order to determine if and which model or estimate may be most reliable for the ecosystem services assessment.

Figure 2: Map of the distribution of benthic production estimates from Cahoon (1999)



Policy Site: Columbia River Estuary

Map showing the distribution of climatic regime (Tropical=60, Temperature=30, Polar=0) and depth (m) across various locations. The radial axis represents depth (m) from 0 to 70. The angular axis represents absolute latitude (°) from 0 to 90. The legend indicates: — Depth (m) — Climatic Regime (Tropical=60, Temperature=30, Polar=0) — Absolute Latitude (°).

Locations and regions labeled on the map include:

- Beniguet Sea, AK, USA
- Cote d'Ivoire
- Signy Is., Scotland
- Ythan Estuary, Scotland
- Laholm Bay, Sweden
- Laholm Bay, Sweden
- Wadden Sea, Netherlands
- Wadden Sea, Netherlands
- Wadden Sea, Denmark
- Danish Fjords
- Danish fjord
- Westerschelde, Netherlands
- Vancouver, BC, Canada
- Chapman Cove, WA, USA
- Morenus-Oleron Bay, France
- Gulf of Trieste, Italy
- Bay of Fundy, Canada
- S. New England, USA
- Golfe de Fos, France
- Golfe de Fos, France
- NW Spain
- Massachusetts Bay, USA
- Woods Hole, MA, USA
- Pottery Pond, RI, USA
- Long Pt. Sound, CT, US
- Chesapeake Bay, VA, USA
- NC Shelf, USA
- Newport Estuary, NC, USA
- Bolsa Bay, CA, USA
- Langebaan Estuary, South Africa
- La Jolla, CA, USA
- Gry's Reef, GA, USA
- Sapelo Is., GA, USA
- AL, USA
- Mississippi Sound, MS, USA
- San Antonio Bay, TX, USA
- Long reef, FL, USA
- Caribbean
- GI Barrier Reef, Australia
- Tuamotu Is., F. Polynesia
- Tuamotu Archipelago
- Tuamotu Archipelago
- Tuamotu Archipelago
- Tuamotu Archipelago
- Moorea, F. Polynesia
- Nosy-Be, Madagascar
- Nosy-Be, Madagascar
- Pacific Islands

Figure 4: Exploratory plots of study site data across contextual variables. Benthic micropalgal production estimates as reported by Cahoon (1999) are shown by contextual variable: depth (upper left), absolute latitude and climate regime (lower left). Other variables such as method (lower right) and the distribution of estimates by publication date and method (upper right) are also shown.



Figure 5: One-way ANOVA model results and boxplots of study site data. Means, medians and interquartile ranges of benthic microalgal primary production estimates (Benthic Production) selected from Cahoon (1999) are shown for the variable groups specified by one-way ANOVA models for selected contextual variables: absolute latitude (upper left), climatic regime (upper right), depth (lower left). Summary statistics (lower right) report significant results at the 99% (**) and 95% (*) levels.

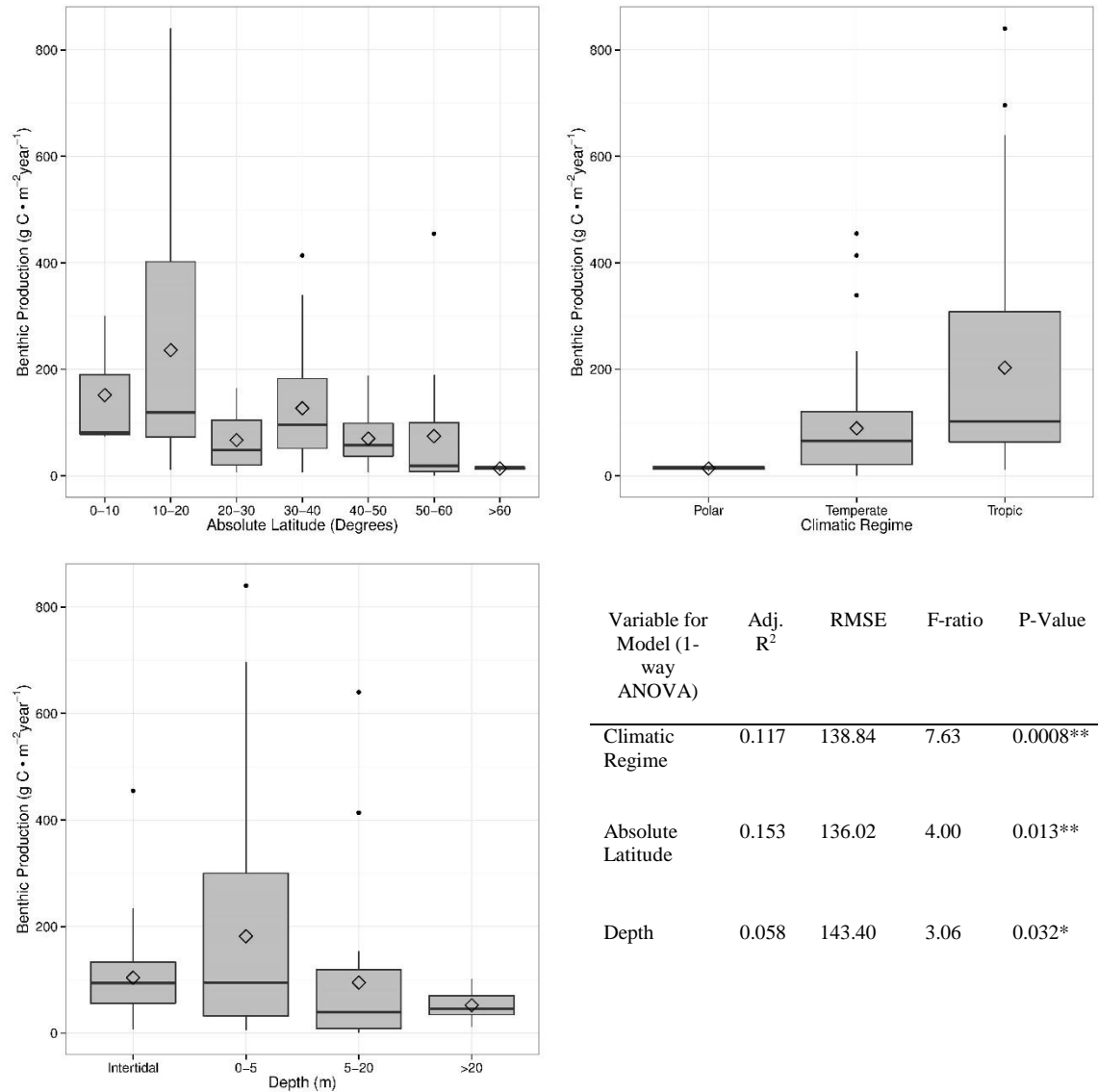


Figure 6: CART Model of benthic microalgal primary production estimates. Splits are defined by the criteria of either absolute latitude or depth variables shown in white boxes. Final nodes (groups) and means are shown in grey.

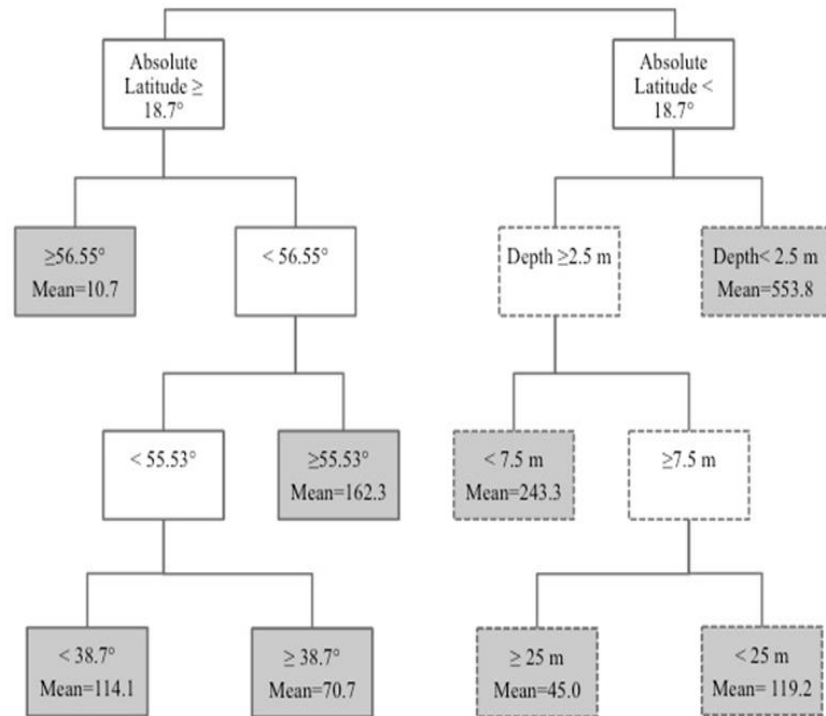
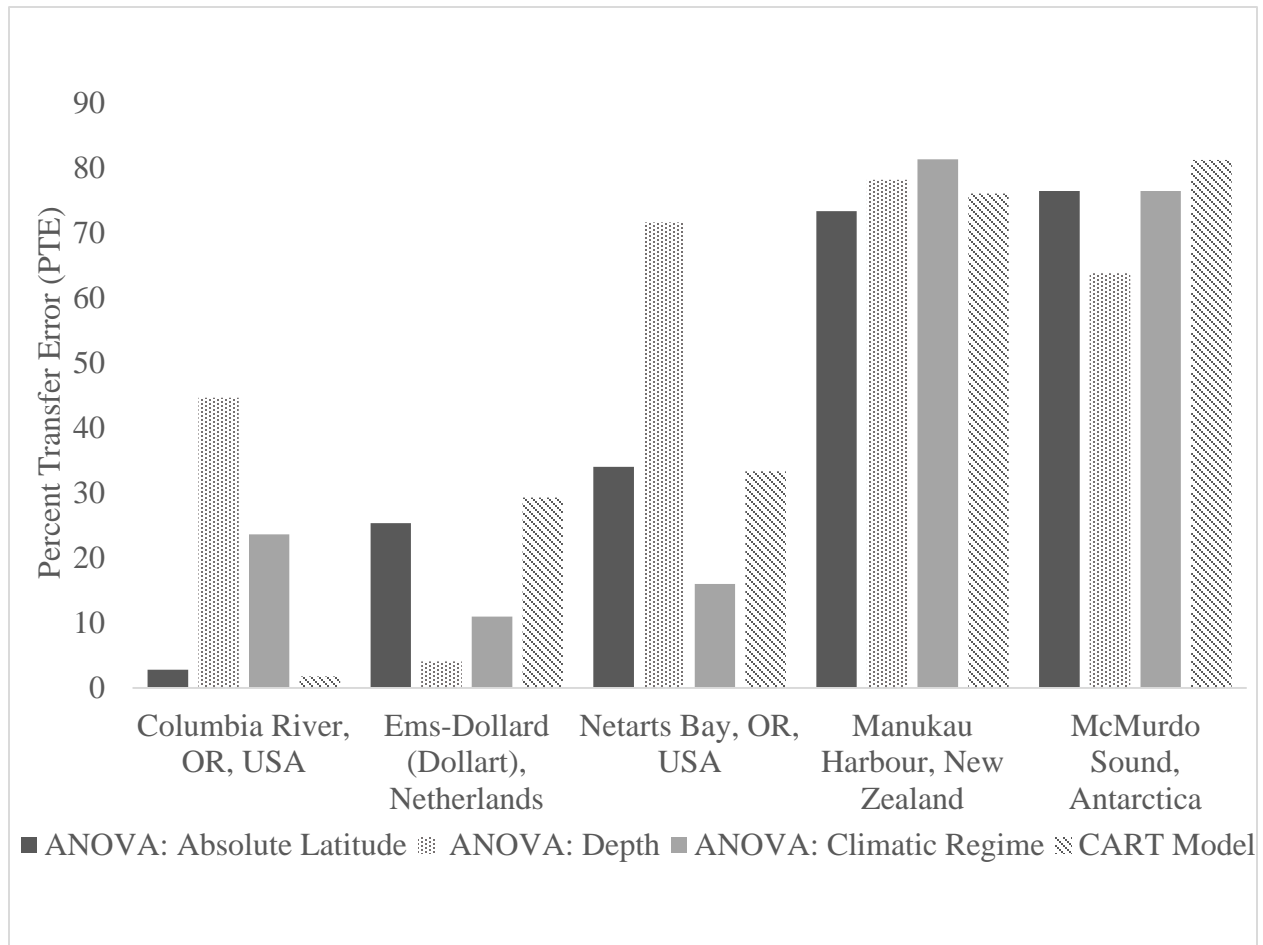


Table 4: Summary policy site model predictions and validation test results. All case study statistical models and an example simple average calculation based on proximate sites are shown. P-values for one-way ANOVA models are shown, CART analysis does not result in a p-value and is therefore not shown. Comparisons of percent transfer error (PTE) are shown for each model predicted mean for two different policy site estimates. The transformed policy site estimate was calculated using the description provided by Cahoon (1999) to adjust hourly benthic microalgal primary production estimates to annual estimates. The true policy site estimate represents the annual value for the site reported by McIntire and Amspoker (1989).

Percent Transfer Error (PTE) Between Study Site and Policy Site Values										
Model or Estimate Type	Context Variable(s)	Mean BMPP (g C m ⁻² yr ⁻¹)	SD	SE	N	R ²	P- value	95% Confidence Interval (g C m ⁻² yr ⁻¹)	Absolute PTE:	
									Transformed Policy Site Estimate	True Policy Site Estimate
ANOVA: Depth	Intertidal	104.13	82.84	13.26	39	0.086	0.032	78.13-130.13	7.0	44.6
ANOVA: Climate Regime	Temperate	89.02	89.19	10.66	70	0.135	0.001	68.13-109.91	20.5	23.6
ANOVA: Absolute Latitude	40-50° Latitude	69.97	45.39	8.90	26	0.203	0.001	52.52-87.42	37.5	2.8
CART	38.7-55.5° Latitude	70.70	91.12	8.74	33	0.597	n/a	53.56-87.84	36.9	1.8
Study Site Average	Washington Sites	78.00	0.82	0.48	3	n/a	n/a	77.07-78.93	30.4	8.3
Average PTE:									26.5	16.2
Transformed Policy Site Estimate	Columbia River Estuary	112								
True Policy Site Estimate	Columbia River Estuary	72								

Figure 7: Validation test results across models. Validation sites were selected from the dataset reported by Cahoon (1999) prior to transferability assessment. Differences between corresponding model predictions based on site characteristics and the value reported by Cahoon (1999) are shown as percentages (PTE). Models shown are (left to right) are one-way ANOVAs for absolute latitude, depth, and climatic regime and the CART model.



6. Discussion:

The case study using benthic microalgal primary production (BMPP) estimates illustrates the breadth of possible sources of transfer errors and the utility of a framework for accounting for these errors. The framework accounts for necessary assumptions using a balanced approach to select estimates for a policy site that meet user-defined accuracy needs. The data set used in the case study has previously been used to support discussion of the challenges and inability to generalize this process at a global scale (Cahoon 2006), and also as the foundation for an ecosystem services approach based on generalizable ecological principles, where declining benthic productivity with depth is one such principle (Townsend et al. 2014). While these

generalizability conclusions are based on the same data, the discrepancy between them helps to illustrate that the generalizability or transferability of information is largely dependent on user requirements for specificity and accuracy and is therefore difficult to define broadly.

The case study assessment provides clear representation of the data and required assumptions if a transfer to the policy site is desired. Ecosystem services research represents a spectrum of policy-pertinent to directly policy-informative research. Therefore, requirements for accuracy and specificity varies within the literature. Because of this, clear and explicit goals, objectives and methods are of paramount importance for ecosystem services studies so that assumptions made within studies are transparently propagated into subsequent studies or policy applications (Costanza et al. 2014; Eigenbrod et al. 2010a,b).

In this section, advantages of the framework will be highlighted in terms of the case study. Afterwards, challenges to applying the framework and for transferability assessment in general will be addressed. In particular, a prominent challenge exists once potential transfer risks are identified, since users must be able to gauge or minimize relative risks.

6.1 Utility of the Ecological Estimate Transfer Framework

The framework presents a logical method for identifying sources of uncertainty and estimating the accuracy of a transferred estimate to assist in the communication of transferability assumptions and risk. Each framework step elucidates sources of uncertainty and assumptions which are recorded and addressed cumulatively to inform a transfer decision. Primary sources of transfer error include generalization and measurement errors, as described by benefit transfer. In order to describe potential sources and indicators of generalization errors, the framework uses

ecological theory to generate a multilevel approach to assess context similarity. The case study helps to support the multilevel representation of generalization error, as represented by the difference between the transformed and true value for the policy site, as well as other sites outside of the dataset. The case study also explores the relative error when different contextual variables are used as the basis for a transfer and finds similarity across climatic regimes and latitudinal gradients to confer the strongest theoretical and statistical evidence to transfer values broadly. In addition, the case study explores statistical and non-statistical approaches that may be used to explore context similarity that may be more accessible to users across disciplines, and for a variety of purposes spanning policy assessment and research.

Users are first required to acknowledge sources of error inherent in the original approach and remove estimates that are inconsistent with the user's standards for conceptual validity. Even when individual estimates are not immediately investigated, as in the example, the potential for error is recorded. Next, users are drawn attention to the sources of error that concern the comparability of estimates, which as illustrated by the case study, may require transformations to maintain a larger dataset. Such adjustments may change the reference frame of the estimates, an indicator of generalization risk. As previously discussed, changes across reference frames change the scale of inference across one or more levels which may affect the nature of the process of interest. In this example, temporal reference frames were adjusted to be comparable and spatial reference frames may have varied from samples within sites (estuaries), representative of entire estuaries, or multiple estuaries in an area.

The transformation from hourly or daily data to annualized estimates as a reference frame change represents the potential for significant error. This error was confirmed by the difference between the hypothetical transformed value and the true estimate for the policy site of 40 g C m^{-2}

hr⁻¹. This bias may be conservative for the magnitude of potential bias since the transformed estimate at least represented an entire years' worth of hourly data. For other estimates, the correction for 10 hour production days and number of production days per year based on climatic regime does not account for when in time the estimate was observed. For example, one of the study site estimates from Massachusetts Bay (Cahoon et al. 1993) derived average hourly BMPP from observations over the course of one week in August. Therefore, the transformation does not account for other seasonal variability and equates the mean from a week in August to the mean over all other months (or 270 days of the year, in temperate zones) that production is assumed to occur. Therefore, processes which affect not only light availability and intensity, but also the presence and abundance of microalgae over longer time scales are not represented by this estimate. For example, light intensity and availability from the sun varies seasonally, as does storm intensity and frequency. Similarly, the extent of observations in Cahoon et al. (1993) was limited to three sites along Stellwagen Bank, within Massachusetts Bay, as opposed to the entire bay as indicated by the location name.

If users permit reference frame assumptions, contextual variables may be used to assess contextual differences in step three. Limitations due to both the *type* and *nature* of contextual variables were acknowledged at this step. Types of contextual variables and their connection to core drivers indicate how well the variables will represent differences between sites that are known to drive the estimate's value. In the case study, reported contextual variables were limited, but seemed to correspond well as proxies of main theoretical drivers.

Main theoretical drivers were identified from literature review in step one to be light, biomass and temperature. Depth serves a proxy of light, because light attenuates with depth (Kauer et al. 2010), absolute latitude and location also serve as proxies for light availability and

temperature, in both intensity and seasonality (Macintyre, Geider, and Miller 1996). Biomass and primary production are dependent, since as one inevitably increases the other; however, neither direct metrics of either abundance, microalgal film density (Santos, Castel, and Souzasantos 1997) or proxies such as grazing pressure (Macintyre, Geider, and Miller 1996) were available. Therefore, users that are constrained by the available intrinsic or extrinsic variables are compelled to acknowledge this limitation.

The effect of using proxies versus direct core variables on the case study assessment was not directly assessed, for lack of data. However, the variability of the data across contextual groups, poor model fits, and problems to predict outside of the dataset may indicate either the inherent variability of the data, or that different variables would represent differences better. Cahoon (1999) also observed the variability in the data across climatic regimes and depth ranges and attributes this to inherent patchiness in observations, differences in between or within locations (light flux), differences between communities (P vs I curves, or photosynthesis versus irradiance responses) and the effect of different measurement techniques.

“Probable sources of variability include patchiness on all scales of time and space, variability in incident light flux and in P versus I responses, and to some extent the different methodologies employed” (Cahoon 1999, p.60)

The nature, or ranges of contextual variables is important for determining how well the pool of study site estimates represents the context of the policy site as an indicator of transfer risk. This was determined through the application niche of estimates, as represented by the map and radar plot of the data. It was determined that considerable variability and patchiness across all variables exists. However, general representation across contextual variables including similar

depths, latitude ranges, and locations of the policy site may reduce potential transfer errors since the transfer location was broadly represented by the data. The validation tests support the importance of representation in the dataset since all models performed worse at the New Zealand and Antarctic sites. While the absolute latitude range, depth range, and climatic regime (temperate) was well represented for the New Zealand estimate, the models performed very poorly for this site. The lack of other geographically proximate estimates prohibit investigation into whether the estimate is anomalous, or if the specific region, estuary, or ecosystem tends to have higher BMPP rates than other sites at similar latitudes. The ANOVA models generally performed slightly better than the CART model across the five validation sites, however CART model performed best for the true policy site estimate (figure 7).

The case study highlights accessible and applicable tools useful for a range of transferability assessment needs. Simple contextual comparisons and analyses can help elucidate the most robust basis for a transfer which balances theoretical guidance and empirical support. Qualitative explorations of the data (e.g., scatterplots and radar plots) identified climatic regime and absolute latitude as the clearest basis to differentiate the context of the policy site from the rest of the dataset whereas quantitative assessment added statistical rigor to these conclusions.

Simple and accessible quantitative tools were also used in the case study. Such methods are advantageous for applications in policy or other situations where time and expertise necessary to build other types of transfer models or conduct full meta-analyses are limited. CART analysis, while potentially more difficult to initially code and set up, is inherently nonparametric and therefore confers several advantages. Firstly, unlike ANOVA, assumptions do not need to be investigated regarding the distributions of the values or the predictors (Lewis 2000). Secondly, a variety of types of data can be used in CART analysis including numerical

data that are highly skewed, categorical or either ordinal or non-ordinal (Lewis 2000; McCune, Grace and Urban 2002). CART results can also be simply visualized in a dichotomously branched tree. ANOVA, by contrast is a more familiar technique that results in a p-value as a widely-used measure of statistical significance. ANOVA also requires groups to be defined by the user. In the case study, 10 degree latitude breaks were used to define groups in the ANOVA model were validated for the policy site, however, for other splits smaller and larger breaks were defined by CART model. This reflects the data-driven and multivariate nature of the CART model which therefore is more robust to user decisions in group selection (i.e., other assumptions concerning context similarity).

Finally, users synthesize both qualitative and quantitative uncertainties to gauge total transfer error in final framework steps. Major sources of uncertainty highlighted in the case study were reference frame changes, contextual variable representation, and statistical uncertainty. In acknowledgement of potential limitations, some transferability conclusions could be tentatively made. The case study benefited from the availability of a ‘true’ estimate for the policy site which allowed transfer error to be determined across the models. The CART model and the absolute latitude ANOVA model were found to minimize PTE at the policy site (1.8 and 2.8 PTE, respectively). The reliance of both models on absolute latitude suggests this contextual variable is the strongest basis to transfer estimates for the dataset and provided variables. This result indicates that both a more complex data-driven classification and a more simple assessment perform comparably for the case study transfer. However, it is also of interest how a transfer could be performed if no policy site information was available. The CART model prediction for the policy site has the lowest standard error ($SE = 8.74$) and the best fit to the data as a whole ($R^2 = .597$) so if a transferability assessment which confirms a strong basis to determine a transfer

estimate. However, the standard error information provided by the model is likely not sufficient to capture the other assumptions and limitations of the dataset.

More detailed uncertainty and validation testing is required to understand total transfer error in order to specify a more reliable error and confidence interval for the estimate. A more conservative confidence interval for the transfer estimate may be prudent, as illustrated by the magnitudes of potential error explored in the validation exercise and resulting from the temporal transformation.

The case study is successful in its description and exploration of the transferability of BMPP estimates. On a broad level, absolute latitude was found to be useful as a basis for determining contextual similarity versus the other contextual variables (climatic regime or depth gradient). Quantitative assessment results show reduced potential gains for more sophisticated transfer models in the presence of broader data-wide limitations. Statistical analysis may assist users to balance theoretical guidance with statistical significance. However, the limitations and assumptions with respect to the contextual reference frames, the available variables, and patchiness of the data confound the ability to determine precise and reliable transfer estimates without the application of more sophisticated analysis.

6.2 Challenges and Future Research Needs

The framework's approach for identifying sources of transfer error potentially confers major advantages for describing assumptions and limitations comprehensively, however major obstacles remain. Broadly, this framework first relies on the importance of outlining the contextual reference frames of policy and study sites. Secondly, intrinsic and extrinsic contextual

variables that relate to the process specific to the reference frame must be specified and related to candidate transfer estimates. Finally, limitations with respect to reference frames or contextual variables must be compared to or combined with other measures of statistical error to inform a final transfer decision. Challenges with respect to these features are discussed next alongside avenues for future research that will support future transferability assessment. Finally, these challenges will be discussed in terms of lessons learned from benefit transfer and recommendations for future research.

6.2.1 Reference Frame Consistency and Evaluation

The extent of ecological theory and research concerning the effect of scale on pattern and process serves as conceptual basis of the contextual reference frame. Furthermore, transfers across the same or similar reference frames reduce the risk of scale-dependent effects or the need to specify scaling relationships. Consistency across reference frames as an indicator of system similarity therefore is a first determinant of transferability, where consistency across intrinsic and extrinsic variables follows closely. This was identified in the case study when a potential inconsistency across reference frames of original studies was identified, along temporal scales, since it required adjustment to be comparable. Consistency across other components of the reference frame was also unclear. Spatial scale in terms of either extent or grain was not reported alongside individual estimates by Cahoon (1999), however it is clear that by definition measures of benthic productivity are at the community level. While the levels of scale that were available for the case study corresponded well with hierarchical levels suggested by the framework (table 2), this may not always be the case for other applications.

Quantifying contextual differences across levels of scale requires users to specify informative levels for the process and estimate in question. The categories of scale (space, time, and ecological organization) are fixed within the ecological scaling literature, however within each category, each hierarchy of scale levels remains general. For example, ecological organization is commonly represented by the levels moving from individual to biome (see table 2; Allen and Starr 1982; Schneider 2001; Wiens 1989; Wu, Jones, and Li 2006), however this hierarchy is not comprehensive. For example, a meta-population is a clearly distinct organizational unit that may not adequately be represented by either the population or community level given by the typical ecological hierarchy presented here. General levels of spatial scale and temporal scale are clear, moving from local to global, and reference to these scales is common however standardization does not exist. In addition, conventions for naming or “classifying” reference frame locations are not always consistent. For example, numerous ecological and biophysical classification systems exist to name scale levels and locations (table 3). Depending on the classification system used, the name of the same location may vary slightly. In addition, since not all classification systems apply globally, a classification designed and applied only to Australia systems may not be easily compared to a system in North America. Therefore, considerable confusion may occur when identifying scale mismatches if different classification systems are used.

Reconciling scale differences for many ecosystem services studies is a prominent challenge because the scales of ecological processes, experiments and analyses may vary from the scales of policy making and management (Wu, Jones and Li 2006). If such scale transitions are common, further research into the consequences, adjustments, or need for more sophisticated modeling may be necessary. A focus of future studies should be the relative consequences across

spatial, temporal and organizational scales and to identify important scale breaks for commonly transferred estimates and services.

6.2.2 Intrinsic and Extrinsic Variable Identification

The example assessment relied primarily on proxies of intrinsic variables, which limits potential accuracy and precision of predicted estimates in a transfer model. Observed variability and limitations of the case study models may be partially due to the lack of informative intrinsic and extrinsic variables. Further transferability investigations may benefit from the use of unadjusted data as response data and more representative intrinsic variables, such as site specific data concerning light intensity and availability across depths and scales. Improved reporting of contextual conditions and descriptions will help improve transferability assessment. In the lack of contextual information within papers, the growing number of classification systems and GIS-referenced databases of biophysical data may permit users to gather appropriate contextual information (e.g., Marine Regions, Claus et al. 2015).

Several ecological classification systems were explored as other potential extrinsic variables in the case study. It was anticipated that ecological classifications could be useful as proxies of differences in ecological composition, salinity regimes, or temperature regimes that were not able to be represented directly by intrinsic variables identified during literature review (Cahoon 1999; Cahoon 2006; Macintyre, Geider, and Miller 1996). In addition, other systematic variations may be represented such as in light availability due to benthic sediment composition, turbidity as related to the site or storm frequency or any number of other covariates are associated with microalgal production estimates but unobserved in many original studies (Cahoon 1999). However, the size and distribution of the case study dataset did not permit the

inclusion of available classifications, such as the MEOW Realms or World Seas, because there was not sufficient replication across groups. These classifications could have been investigated if assessment was limited to the temperate observations of the dataset, since many unique observations were across tropical or polar locations.

Their efficacy of available classifications for describing meaningful differences between estimates depends on the basis of each system and the ability to apply the classification (Troy and Wilson 2006). This is particularly true when classification systems are limited in scope to particular continents or regions which cannot be broadly compared. In assistance to this end, several hierarchical and global classification systems have been proposed for marine systems.

Marine Ecoregions of the World (MEOW, Spalding et al. 2007) and the Coastal and Marine Ecological Classification Standard (CMECS, NOAA 2012) represent two promising classification systems for use in transferability assessment of coastal ecosystem service estimates. CMECS classifies environmental “units” (a flexible description given to multiple scales) based on two settings (both aquatic and biogeographic) and four components (water column, geoform, substrate, and biotic). CMECS utilizes the MEOW classification to describe biogeographic setting. MEOW identifies coastal settings using a three-tiered hierarchical approach moving from realms (largest) to provinces to ecoregions (smallest). Given the size of our dataset and the spatial resolution of studies only realms were able to be georeferenced to the case study data. The MEOW classification is driven primarily to discriminate between distinct biota for conservation of biodiversity globally, however Spalding et al. (2007) notes the relationship and correspondence between biotic relationships and abiotic drivers:

“Very large regions of coastal, benthic, or pelagic ocean across which biotas are internally coherent at higher taxonomic levels, as a result of a shared and unique evolutionary history. Realms have high levels of endemism, including unique taxa at generic and family levels in some groups. Driving factors behind the development of such unique biotas include water temperature, historical and broad scale isolation, and the proximity of the benthos.” (Spalding et al. 2007, p. 3)

Therefore, finer, more descriptive variations among studies may correlate with some important abiotic drivers or species compositions. However, the unevenly and unequitable distributed estimates in the case study disallowed reliable use of this biogeographic classification. Classification systems which apply broadly that relate characteristics which vary quantitatively across sites may be better able to describe differences across geographically distributed and sparse data. CMECS represents one such classification system. For example, the water column component is further defined by water layers (e.g., surface layer, upper water layer, pycnocline, and lower water layer), temperature, salinity, hydroform, and biogeochemical features (NOAA, 2012). Therefore, as studies adopt and relate individual studies to this classification, the amount of available information for transferability assessment will be greatly improved. As recognition and adoption of the CMECS standard grows, transferability assessment across possible intrinsic and extrinsic variables will be greatly enabled and will further reduce dependencies on proxies.

6.2.3 Quantitative Estimates of Total Transfer Error:

The generalizations and simplifications of the dataset prior to analysis reduce the ability for the transfer models to predict reliable policy site estimates and for statistical error measures

to represent the uncertainty of the models to perform outside of the dataset. Therefore, either additional analyses quantifying the effect of these errors or decision rules on how estimate error should be adjusted are needed.

The case study provided cursory assessment of the potential magnitude of these errors through the validation testing and the hypothetical transformation of the policy site estimate. Neither is sufficient to specify a new confidence interval for the transfer estimate, but these results do indicate several issues. Firstly, the hypothetical transformed policy site estimate value still lies within one standard deviation of the mean, which indicates the confidence interval and error may be conservative enough to encapsulate this error for the policy site. However, as previously mentioned, the deviation between the true policy site estimate and the transformed value may be conservative and therefore not representative of this error across the entire dataset. Secondly, PTEs across validation sites vary widely. The PTEs observed for the New Zealand and Antarctic sites reflect that the true estimates do not lie within two standard deviations of the predicted means, regardless of the model used. The improved performance of the models for sites similar to the policy site confers some confidence for the models for these conditions, however overall this indicates the need for further investigation of the data. Further investigation may specify more accurate or representative values of sites, more potential predictors, and comparisons across models using various specifications.

Additional tools and methods may assist with transferability assessment. In particular, spatial statistical tools such as empirical variograms and ordination plots may help to investigate distance related effects across estimate locations. Literature review found that geographic proximity is a common basis used for transfers however, this was only indirectly evaluated by the case study. Using measures of distance between georeferenced points, points in time, or

ecological relationships variability in estimates as a function of distance, these relationships can be evaluated (Isaaks and Srivastava 1989; Legendre, 1993).

6.2.4 Insights from Benefit Transfer and Future Research Directions

Because of the previously identified challenges to quantify and evaluate context similarity there are fundamental challenges and limitations for conducting robust ecological estimate transfers. Benefit transfer has similarly noted such obstacles and through repeated testing across contexts and location has derived a few general conclusions. To follow this example, future research should focus efforts on quantifying trade-offs between methods for conducting transfers, prioritizing transferability research on estimates that are in greatest limitation or demand, and deriving general guidance for the relative importance of different contextual reference frame components. In particular, broad indicators of context are important if robust point estimate transfers are to be made.

The benefit transfer literature has generally found that function-based transfers outperform unit value transfers (Rosenberger and Stanley 2006), where function transfer refers to the suite of function-based transfer approaches including meta-analysis models and others (see Johnston and Rosenberger 2010 for other methods). In the case study it was shown how quantitative methods including ANOVA and CART analysis can inform transferability assessment that balances available information and user knowledge and time constraints. However, comparisons between such methods and more detailed function-based transfers and meta-analytic models will help inform when simple approaches for estimate transfer, such as those shown are sufficient as compared to other approaches.

As has occurred in the benefit transfer literature, quantitative investigations of transfer error with respect to specific services and contexts will yield more specific transferability guidance and may modify, add or emphasize the importance of contextual reference frames and contextual variables. The presented framework applies a conservative approach to identifying risks associated with differences in reference frames, however ecosystem-service specific investigations into the magnitude of error introduced by changes in reference frames, both to predicted estimates in the transferability exercise and to final ecosystem service model results are needed to fully gauge the importance of similarity across reference frames. In addition, such studies will inform the importance for extrinsic and intrinsic variables in transfer models, which can in turn help to guide conceptual or qualitative assessments.

Suggestions for improving benefit transfer have included increasing the availability of original data and reporting site characteristics and study attributes more consistently (Loomis and Rosenberger 2006). Reporting of site characteristics consistent with the contextual reference frame will assist scale identification and comparison. In addition, this will facilitate the identification of relevant extrinsic and intrinsic variables. Availability of original data may also assist in variable identification and concurrently facilitate meta-analytic analyses.

Databases such as the Environmental Valuation Resource Inventory (EVRI) have been designed in part in order to enhance benefit transfer practices (DeCivita et al. 1998). Similarly, the EcoService Model Library (ESML), a database of ecological models, seeks to collect information on ecological models that relate to ecosystem service production (EPA, 2015). This database records detailed information that may help users identify equivalent scales and contextual variables. For example, the database collects information on model extent, grain, ecological organization in both numerical and categorical descriptions. Point locations in space

and in time are recorded for the model in general, and information on the lag interval and distribution of observations, if temporally or spatially distributed, is also recorded. Descriptive names are applied to models as well as they correspond to the author's stated spatial scale are provided, but standardized classification schemes are also used for both terrestrial (using continent, country, and state identifiers) and marine locations (Marine Ecoregions of the World, MEOW). Such a descriptive tool may aid in the identification of potentially useful ecosystem service estimates or models for transfer.

More extensive reviews of transfer practices within the ecosystem services literature will allow the prioritization of future transferability analyses. In this thesis, a focused review of the habitat-fisheries valuation literature found that estimates of natural mortality rate and primary production were transferred most often. A more comprehensive review of the literature and of policy and management applications will help identify the most commonly transferred estimates. Future discussion and self-identification of the use of ecological estimate transfer, as occurs with benefit transfer, in the literature will help improve the ability to review the occurrence of the practice and in turn prioritize areas in most need of detailed transferability assessment.

The case study is effective in highlighting important determinants of transfer error and provides several indicators for the magnitude of these errors, however the case study is limited in both replication and scope. More comprehensive analysis is needed in order to understand the relative effects of generalization errors due to differences across contextual features such as the reference frame levels, locations and variables. The validation estimates used in the case study illustrate some trade-offs across the models for predicting at sites with different characteristics but more intensive cross-validation techniques would permit more concrete conclusions concerning relative model performance. In addition, by comparing models built using either

transformed or untransformed data the magnitude of this transformation on model predictions could be better assessed. Finally, rebuilding models using different combinations of extrinsic and intrinsic variables and utilizing the study site data more comprehensively will help define more meaningful groups and reduce statistical sources of uncertainty.

7. Conclusion

Demand for ecosystem service assessment is growing, from both an academic and a policy perspective. The concept of ecosystem services as a tenant of ecosystem-based management approaches is growing in importance to agencies, such as the U.S. EPA (EPA 2012) and has been written into the language of the U.S. National Ocean Policy (*National Ocean Policy Implementation Plan* 2013). Furthermore, ecosystem service assessment has been directly requested as part of the natural resource damage assessment for the Deepwater Horizon oil spill (Mayer et al. 2012). Regulatory applications of ecosystem services information may increase the demand for ecological estimate transfers as has been observed for benefit transfer (Bergstrom and DeCivita 1999). Comprehensive ecosystem service assessments require researchers to extract information potentially across diverse ecological disciplines and sub-disciplines (Börger et al. 2014; Guerry et al. 2013). The frequency of transfers in the habitat-fisheries valuation literature is an indicator of the occurrence of the practice, especially in methods which utilize production functions and rely on more ecological information. If ecosystem service research and values are to become management and policy informative, transparency and effective communication of limitations and assumptions is requisite (EPA 2009). The presented framework and associated structure for defining ecological contexts, the

contextual reference frame, represents a first attempt to organize and standardize transfers of ecological information in ecosystem services research. This approach represents a synthesis of previous work and thought in economic benefit transfer, ecological scaling, and ecosystem services.

Benefit transfer as an analog demonstrated the theoretical importance of context for driving transfer errors and helped to define an architecture of ecological contexts. Previous work has identified the shortcomings of some types of transfers, most notably generalizations across spatial scale (Eigenbrod et al. 2010b). The framework draws attention to the multidimensionality of transfers across scale and contextual variables and calls for future research comparing the relative consequences of transfers that occur across different contextual reference frame scales, levels and locations. In particular, the case study drew attention to generalization errors across temporal scales. The proposed structure for defining context relates to the need to identify appropriate levels of aggregation and linking information across scales in ecology as a whole (Meyer et al. 2010) as well as the creating of functionally meaningful classifications of context for ecosystem services valuation (Troy and Wilson 2006). The structure for ecological contexts proposed in this thesis serves as a baseline for gauging transferability risk, but future research is needed to understand the relative risks between generalizations across spatial, temporal and ecological scales. The presented framework emphasizes the importance of diligence in each step of the meta-analytic and research synthesis process, asking researchers to define and compare the transfer needs to the available data, not only to assure consistency and accuracy underlying each candidate estimate itself, but also to compare estimates in terms of their contexts.

The case study explored the advantages and challenges for comprehensive description of potential transfer errors. CART and ANOVA analyses are potentially useful methods to explore

ecological estimate transfers in addition to the meta-analytic and function-based approaches used in benefit transfer (Bergstrom and Taylor 2006). These methods allow comparisons between estimates and contexts and to assess transfer risks the need for further transferability assessment. The case study analysis highlights the utility of these approaches and the potential errors for generalizations across temporal scales. Given sufficient data, trends and effects across dimensions and metrics of contexts may be better understood in order to inform decisions concerning the generalizability of estimates or values for a given scale. Other techniques as applied in the spatial statistical literature, such as empirical variograms, hold promise for future transferability assessment across spatial locations (Legendre 1993; Pigott 2012).

Considerable work is needed to quantify and compare magnitudes of transfer error for transfers occurring across contextual reference frames and locations to generate better indicators of transfer reliability. Indicators of transfer reliability that are less computationally demanding than full meta-analyses, or that may be used when too little information is available for comprehensive assessment will be useful but first depend on generating a body of supporting transferability assessments. If parameters or estimates that are most widely transferred can be identified, subsequent work may first focus on these estimates. In addition, investigations into the validity of traditional ecosystem classes, habitat types, or LULC as organizationally useful metrics of transferability should be conducted to test the extent of utility of preexisting classifications for transferability assessment.

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Appendix 1: Metadata used for case study analysis. Data reproduced with publisher permission from table 3 in Cahoon (1999) includes the benthic microalgal primary production (BMPP) estimate, location, median depth, method, publication year and climatic regime. Google Earth and ArcGIS were used to identify the latitude, longitude, World Sea, and MEOW Realm information corresponding to the location name provided by Cahoon (1999). Estimates not applicable for analysis

Location	BMPP (g C m ⁻² yr ⁻¹)	Depth Binned (median)	Method	Publication Year	Climatic Regime	Absolute Latitude (°)	Longitude (°)	World Sea	Realm (MEOW)
AL, USA	20	1.2	14C	1995	Te	30.71	30.71	-86.9 Gulf of Mexico	Temperate Northern Atlantic
Barataria Bay LA, USA	121	0	O2	1988	Te	29.39	29.39	-89.94 Gulf of Mexico	Temperate Northern Atlantic
Bay of Fundy, Canada	58	0	O2	1983	Te	44.77	44.77	-65.38 Bay of Fundy	Temperate Northern Atlantic
Beaufort Sea, AK, USA	9	7	14C	1982	Po	72.84	72.84	-145.41 Beaufort Sea	Arctic
Boca Ciega, FL, USA	57	2.5	O2	1960	Tr	27.8	27.8	-82.76 Gulf of Mexico	Tropical Atlantic
Bolsa Bay, CA, USA	97	0	14C	1978	Te	33.7	33.7	-118.05 North Pacific Ocean	Temperate Northern Pacific
Carribean	40.9	33.5	O2	1972	Tr	21.47	21.47	-78.65 Caribbean Sea	Tropical Atlantic
Chapman Cove, WA, USA	67	0	O2	1994	Te	47.22	47.22	-123.03 The Coastal Waters of Southeast Alaska and British Columbia	Temperate Northern Pacific

Chesapeake Bay, VA, USA	234	0 O2	1985	Te	37.58	37.58	-76.17	North Atlantic Ocean	Temperate Northern Atlantic
Chesapeake Bay, VA, USA	196	1 O2	1985	Te	37.584	37.584	-76.17	North Atlantic Ocean	Temperate Northern Atlantic
Chesapeake Bay, VA, USA	75	0 14C	1993	Te	37.58	37.58	-76.17	North Atlantic Ocean	Temperate Northern Atlantic
Chukchi Sea, AK, USA	16	5 14C	1974	Po	69.47	69.47	-171.69	Chukchi Sea	Arctic Tropical
Cote d'Ivoire	81	2.5 O2	1977	Tr	5.13	5.13	-5.36	Gulf of Guinea	Atlantic Tropical
Curacao	405	1.75 O2	1976	Tr	12.17	12.17	-68.99	Caribbean Sea	Atlantic Tropical
Danish fjord	15	0.5 14C	1980	Te	54.59	54.59	8.42	North Sea	Atlantic Temperate Northern
Danish fjord	5	6 14C	1980	Te	54.59	54.59	8.42	North Sea	Atlantic Temperate Northern
Danish Fjords Ems-Dollard (Dollart), Netherlands	189	0.9 14C	1960	Te	54.59	54.59	8.42	North Sea	Atlantic Temperate Northern
	132	0 O2	1982	Te	53.28	53.28	7.18	North Sea The Coastal Waters of Southeast Alaska and British Columbia	Temperate Northern Pacific
False Bay, WA, USA	108	0 O2	1968	Te	48.48	48.48	-123.068	Mediterranean Sea - Western Basin	Temperate Northern Atlantic
Golf de Fos, France	54	0.5 O2	1984	Te	43.38	43.38	4.93		

Golfe de Fos, France	32	3.75 O2	1994	Te	43.38	43.38	4.93	Mediterranean Sea - Western Basin Mediterranean	Temperate Northern Atlantic
Golfe de Fos, France	100	1 O2	1996	Te	43.38	43.38	4.93	Mediterranean Sea - Western Basin Mediterranean	Temperate Northern Atlantic
Golfe de Fos, France	26.5	11 14C	1996	Te	43.38	43.38	4.93	Mediterranean Sea - Western Basin Mediterranean	Temperate Northern Atlantic
Golfe de Fos, France	54	0.5 O2	1987	Te	43.38	43.38	4.93	Mediterranean Sea - Western Basin North Pacific	Temperate Northern Atlantic
Gray's harbor, WA, USA	59	0 O2	1984	Te	46.93	46.93	-123.7	Ocean North Atlantic	Temperate Northern Pacific
Gray's Reef, GA, USA	120	20 O2	1991	Te	31.98	31.98	-80.93	Ocean	Temperate Northern Atlantic
Great Barrier Reef, Gt. Barrier Reef,	696	0.5 O2/14C	1985	Tr	18.28	-18.28	147.7	Coral Sea	Central Indo-Pacific
Reef,	333	4 O2	1990	Tr	18.28	-18.28	147.7	Coral Sea	Central Indo-Pacific
Gulf of Trieste, Italy	57	7 O2	1989	Te	45.67	45.67	13.58	Adriatic Sea	Temperate Northern Atlantic
Gulf of Trieste, Italy	40	15 O2	1989	Te	45.67	45.67	13.58	Adriatic Sea	Temperate Northern Atlantic
Gulf of Trieste, Italy	30	22 O2	1989	Te	45.67	45.67	13.58	Adriatic Sea	Temperate Northern Atlantic

Hokkaido, Japan	19	0	Ixchla	1991	Te	43.06	43.06	141.34	Japan Sea	Temperate
La Jolla, CA, USA	414	18.3	O2	1978	Te	32.83	32.83	-117.27	North Pacific	Northern
Laguna de Terminos, Mexico	73	1.75	O2	1982	Tr	18.7	18.7	-91.56	Gulf of Mexico	Temperate
Laholm Bay, Sweden	8	11.5	O2	1986	Te	56.55	56.55	12.94	Kattegat	Northern
Laholm Bay, Sweden	15	15	O2	1986	Te	56.55	56.55	12.94	Kattegat	Atlantic
Laholm Bay, Sweden	3.5	17.5	O2	1986	Te	56.55	56.55	12.94	Kattegat	Temperate
Laholm Bay, Sweden	1	19.5	O2	1986	Te	56.55	56.55	12.94	Kattegat	Northern
Laholm Bay, Sweden	18.5	3.5	14C	1988	Te	56.55	56.55	12.94	Kattegat	Atlantic
Laholm Bay, Sweden	7	6.5	O2	1986	Te	56.55	56.55	12.94	Kattegat	Temperate
Langebaan Estuary, South Africa	7	0	14C	1988	Te	33.09	-33.09	18.03	South Atlantic Ocean	Northern
Loch Ewe, Scotland	5.8	4	14C	1969	Te	57.82	57.82	-5.63	Inner Seas off the West Coast of Scotland	Temperate
Long Is. Sound CT, US	41	0	14C	1986	Te	41.07	41.07	-72.94	North Atlantic Ocean	Northern

Long Is. Sound, NY, USA	6.6	0.9 14C	1996	Te	41.07	41.07	-72.94	North Atlantic Ocean	Temperate Northern Atlantic
Long reef, FL, USA	13.5	16 ND	1972	Tr	25.42	25.42	-80.126	North Atlantic	Tropical Atlantic
Marennes- Oleron Bay, France	127	0 O2	1994	Te	45.8	45.8	-1.1	Bay of Biscay North	Temperate Northern Atlantic
Marshall Is	300	2.5 O2	1973	Tr	7.23	7.23	171.18	Pacific North Atlantic	Eastern Indo-Pacific
Massachusetts Bay, USA	36	24.5 O2	1993	Te	42.32	42.32	-70.78	Ocean	Temperate Northern Atlantic
Mississippi Sound, MS, USA	339	1 14C	1992	Te	30.27	30.27	-88.51	Gulf of Mexico	Temperate Northern Atlantic
Mississippi Sound, MS, USA	63	0 14C	1988	Te	30.26	30.26	-88.51	Gulf of Mexico South	Temperate Northern Atlantic
Moorea, F. Polynesia	402	0.5 O2	1976	Tr	17.54	-17.54	-149.83	Pacific North Atlantic	Eastern Indo-Pacific
NC Shelf, USA	50.5	17.5 O2	1992	Te	35.25	35.25	-75.43	Ocean North Atlantic	Temperate Northern Atlantic
NC Shelf, USA	94	25 O2	1992	Te	35.25	35.25	-75.43	Ocean North Atlantic	Temperate Northern Atlantic
NC Shelf, USA	55.6	35.5 O2	1992	Te	35.25	35.25	-75.43	Ocean North Atlantic	Temperate Northern Atlantic
Newport Estuary, NC, USA	20	0 14C	1977	Te	34.76	34.76	-76.7	Ocean North Atlantic	Temperate Northern Atlantic
North River, SC, USA	135	0 O2	1993	Te	33.33	33.33	-79.17	Ocean Atlantic	Temperate Northern Atlantic

Nosy-Be, Madagascar	104	5 14C	1978	Tr	13.31	-13.31	48.25	Mozambique e Channel	Western Indo- Pacific
Nosy-Be, Madagascar	85	15 14C	1978	Tr	13.31	-13.31	48.25	Mozambique e Channel	Western Indo- Pacific
Nosy-Be, Madagascar	42	25 14C	1978	Tr	13.31	-13.31	48.25	Mozambique e Channel	Western Indo- Pacific
Nosy-Be, Madagascar	17	35 14C	1978	Tr	13.31	-13.31	48.25	Mozambique e Channel	Western Indo- Pacific
Nosy-Be, Madagascar	12	60 14C	1978	Tr	13.31	-13.31	48.25	Mozambique e Channel	Western Indo- Pacific
Nova Scotia, Canada	19	0 O2	1986	Te	44.68	44.68	-63.74	North Atlantic Ocean	Temperate Northern Atlantic
NW Spain	79	0 14C	1985	Te	43.06	43.06	-9.25	North Atlantic Ocean	Temperate Northern Atlantic
Oresund, Denmark	95	1 14C	1970	Te	55.75	55.75	12.68	Kattegat Solomon	Central Indo- Pacific
Pacific Islands	74	5.5 ND	1986	Tr	7.93	-7.93	157.57	Sea North Atlantic	Temperate Northern Atlantic
Potters Pond, RI, USA	150	0.7 O2	1985	Te	41.4	41.4	-71.53	Ocean North Pacific	Temperate Northern Pacific
S. Cal lagoon, CA, USA	143	0 O2	1983	Te	34.1	34.1	-119.08	Ocean North Atlantic	Temperate Northern Pacific
S. New England, USA	54	0 O2	1971	Te	43.4	43.4	-70.39	Ocean North Atlantic	Temperate Northern Atlantic
Salt marsh, DE, USA	74	0 O2	1974	Te	38.7	38.7	-75.16	Ocean	Temperate Northern Atlantic
San Antonio Bay, TX, USA	7.3	2 IxChla	1996	Te	28.3	28.3	-96.71	Gulf of Mexico	Northern Atlantic

Santos Estuary, Brazil	164	0 14C	1983	Tr	23.97	-23.97	-46.33	South Atlantic Ocean North Atlantic	Temperate South America Temperate Northern
Sapelo Is., GA, USA	25	0 14C	1981	Te	31.42	31.42	-81.24	Ocean North Atlantic	Atlantic Temperate Northern
Sapelo Is., GA, USA	214	0 14C	1976	Te	31.42	31.42	-81.24	Ocean North Atlantic	Atlantic Temperate Northern
Sapelo Is., Georgia	47	0 O2	1959	Te	31.42	31.42	-81.24	Ocean North Atlantic	Atlantic Temperate Northern
Savin Hill Cove MA, USA	188	0 14C	1990	Te	42.31	42.31	-71.04	Ocean North Atlantic	Atlantic Temperate Northern
SC, USA	136	0 O2	1986	Te	33	33	-79.11	Ocean Indian	Atlantic Central Indo-Pacific
Scott Reefs, Indian Ocean	640	5.5 O2	1986	Tr	14	-14	121.83	Ocean Southern	
Signy Is., S.Orkney Is.	16	8 14C	1991	Po	60.71	-60.71	-45.6	Ocean	Southern Ocean Temperate Northern
Smalandshavet, Denmark	89	4 14C	1972	Te	55.02	55.02	11.51	Kattegat	Atlantic Tropical
St.Croix, Virgin Islands	840	1.75 O2	1985	Tr	17.71	17.71	-64.83	Caribbean Sea	Atlantic Temperate Northern
SW England	143	0 14C	1978	Te	50	50	-5.41	English Channel North Pacific	Atlantic Temperate Northern
Tijuana Bay, CA, USA	232	0 O2	1980	Te	32.56	32.56	-117.09	Ocean South Pacific	Pacific Eastern Indo-Pacific
Tuamotu Archipelago	171	2.5 O2	1988	Tr	18.04	-18.04	-141.41	Pacific South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	154	7.5 O2	1988	Tr	18.04	-18.04	-141.41	Pacific	Pacific

Tuamotu Archipelago	137	12.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	119	17.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	101	22.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	84	27.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	66	32.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Archipelago	49	37.5 O2	1988	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
Tuamotu Is, F. Polynesia	426	0.5 O2	1976	Tr	18.04	-18.04	-141.41	South Pacific	Eastern Indo-Pacific
The Coastal Waters of Southeast Alaska and British Columbia									Temperate Northern Pacific
Vancouver, BC, Canada	106	0 O2	1976	Te	49.26	49.26	-123.11	Columbia	Temperate Northern Pacific
Wadden Sea, Denmark	455	0 14C	1962	Te	55.53	55.53	8.08	North Sea	Temperate Northern Atlantic
Wadden Sea. Netherlands	115	0 O2	1982	Te	55.53	55.53	8.08	North Sea	Temperate Northern Atlantic
Wadden Sea. Netherlands	130	0 14C	1980	Te	55.53	55.53	8.08	North Sea	Temperate Northern Atlantic
Wadden Sea. Netherlands	94	0 14C	1974	Te	55.53	55.53	8.08	North Sea	Temperate Northern Atlantic
Wadden Sea. Netherlands	85	0 14C	1977	Te	55.53	55.53	8.08	North Sea	Temperate Northern Atlantic

Westerschelde , Netherlands	10	0 O2	1995	Te	51.42	51.42	3.54	North Sea	Temperate
Woods Hole, MA, USA	70	0 14C	1974	Te	41.5	41.5	-70.67	North Atlantic	Northern
Woods Hole, MA, USA	95	0 14C	1976	Te	41.5	41.5	-70.67	North Atlantic	Temperate
Ythan Estuary, Scotland	20	0 14C	1970	Te	57.3	57.3	-1.98	Ocean	Northern
Columbia River, OR, USA	72	0 O2	1986	Te	46.24	46.24	-123.8	North Pacific	Atlantic
Ems-Dollard (Dollart), Netherlands	100	0 14C	1984	Te	53.28	53.28	7.18	North Sea	Temperate
Netarts Bay, OR, USA	106	1 O2	1983	Te	45.4	45.4	-123.94	North Pacific	Northern
Manukau Harbour, New Zealand	476	0 IRGA	1981	Te	37	-37	174.66	Ocean	Pacific
McMurdo Sound, Antarctica	58	18 14C	1986	Po	77.84	-77.84	166.67	Southern Ocean	Temperate
									Australasia
									Southern Ocean